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PALCO THE PACIFIC LUMBER COMPANY P.O. Box 37, Scotia, CA 95565 (707) 764-2222

November 16, 1998

RECEIVED

US ... CCFWO, Arcata, CA

Mr. Bruce Halstead
U.S. Fish and Wildlife Service
1125 16th Street, Room 209
Arcata, CA 95521

RE: Pacific Lumber Company, Scotia Pacific Company LLC, et al. ("PALCO") Comments and Responses To Draft Environmental Impact Report/Environmental Impact Statement Reviewing and Evaluating It's HCP/SYP, Etc., Permit Nos. PRT-82895 and 1157

Dear Mr. Halstead:

On behalf of PALCO, we offer the following comments on the draft EIR/EIS, prepared and circulated through the CEQA and NEPA review process and evaluating our HCP/SYP and ITP permit applications, described above.

The various divisions and departments of the Company, and its management, consultants, contractors and counsel have reviewed the draft EIR/EIS, and we prepare and present this document detailing a number of our comments along with substantive information, analysis, or assessment for inclusion in the Administrative Record and for consideration by the permitting agencies in preparing the final EIS, as appropriate.

We have attempted to correct inaccuracies or inconsistencies we note in the draft EIR/EIS, to provide updated or corrected data where appropriate, or to address concerns of which we have taken note during the several public hearings on the draft EIR/EIS and the HCP/SYP. Due to the scope of comments offered during the public hearing process, and the broad nature of the draft environmental review documents, our comments touch upon a number of issues and topics, and we attempt to provide clarity by identifying the topics in bold headings as we discuss them below.

Please respond to these comments, questions, or clarifications in preparation of the final EIR/EIS. Due to the broad range of comments, issues, or questions raised during the review process thus far, our comments do not follow the order or organization presented in the draft EIR/EIS. We provide immediately below a brief synopsis of the topics or issues upon which we comment.

I. Data Clarification, Correction, and/or Comment Where Inaccuracy or Inconsistency Noted in the Draft EIR/EIS

II. Independent Assessments or Analyses By Consultants or Experts Responsive To Public Comments, Concerns, or Expressions of Need For Further Analysis in the Draft EIR/EIS

III. Information, Data or Analysis from PALCO Responsive To Public or Agency Comments Noted Through the Draft EIR/EIS Review Process

I. Data Clarification, Correction, and/or Comment Where Inaccuracy or Inconsistency Noted in the Draft EIR/EIS

The number at the left refers to a specific page in the draft EIS/EIR; a corresponding clarification, correction, or comment is noted immediately to the right. In a few instances, changes that should be made globally are noted.

	Page	Clarification, Correction, and/or Comment
	Introduction:	
PL-1	ix	Figure 2.5-4. Marbled should be caps.
PL-2	xii	Table 3.7-6. What is footnote after title? Table3.7-11. Should be Seral Stage, not State
PL-3	xix	mbf and mbfn are thousand board feet, ***not million***. PALCO…Scotia Pacific Company LLC, not Scotia Pacific Holding Company.
	Summary:	
PL-4	S-3	PALCO also owns a substantial portion of the Freshwater watershed.
PL-5	S-3	It is inappropriate to state that "PALCO's harvest of ***old growth trees*** would likely result in take of listed species." It should be stated that PALCO's harvest of ***timber*** would likely result in take of listed species."
PL-6	S-8	Protection of Headwaters is an important component of the Proposed Action/Proposed Project **and it is mitigation for PALCO's proposed HCP/SYP.**
PL-7	S-11	Table S-1 has at least one error: For Class I streams under Alternative 2, the 170-foot no-harvest RMZ should have a reference to footnote 5.
PL-8	S-13	A WLPZ is a watercourse and lake protection zone (see 14 CCR 895.1).
PL-9	S-14	Table S-2 should contain a footnote that describes how the "Prior Impacts to Watershed" assessments were made. It is our understanding that anecdotal and qualitative information was largely used to make these determinations. It is of interest to note that quantitative data recently obtained by CDF "suggest that only minor channel aggradation may have occurred in the lower gradient reaches of Freshwater Creek." This information is contained in a memorandum from Mr. John Marshall (CDF) to Mr. Dave Ebert (CDF) dated September and is submitted as part of this letter as Attachment I-1.
PL-10	S-15	Table S-3: This table reports low to moderate levels of LWD recruitment expected under Alternative 2 during the interim period. A strong case can be made that this is an underestimate of LWD recruitment expected from the interim mitigation measures given that: 1) the 30 ft. no harvest buffer alone would be expected to provide from 45-80 percent of the LWD

expected from an old growth forest (see in particular McKinley[1] who studied LWD recruitment from second growth forests); 2) Over 75 percent of the THPs that PALCO has modified to the interim measures to date have had 100-170 ft wide no harvest zones along Class Is, and 100 ft. wide no harvest zones along Class IIs; and 3) recently published work from Casper Creek showing that buffers have higher than expected levels of LWD recruitment because of elevated blow down rates (USDA[2]). In addition, the interim measures include, because of AB 1986, no harvest buffers of 100 ft. along all Class I streams, and 30 ft. along all Class II streams. Thus, the interim measures are now more protective for LWD than the default prescriptions for Alternative 2, which you rated as having moderate to high levels of LWD protection for Class I and II streams. Your assessment of the protection level for LWD for the interim measures under Alternative 2 therefore needs to be upgraded. This mistake is repeated throughout the EIS in subsequent sections (e.g., Table 2.6-1). It needs to be corrected throughout the document.

PL-11 S-16 You indicate that the "essential difference between Alternative 1 and the other action alteratives with respect to fish habitat" is that Alt. 1 does not include provisions for road management. However, Alternative 1 also does not include measures to reduce mass wasting as, for example, Alternative 2 does. Given recent sediment source studies on PALCO's lands showing that the overwhelming majority of sediment input to streams on the ownership comes from mass wasting (PWA[3], PWA[4], PWA[5]), the lack of a mass wasting avoidance strategy in Alternative 1 raises serious questions about this alternative's ability to reduce sediment delivery to streams. Despite this Alternative 1 repeatedly receives moderate, high and complete ratings for sediment control (e.g., Table S-3, page S-15). How do you justify and explain these findings for Alternative 1?

PL-12 S-19 The numbers in Table S-6 are dimensioned with the wrong units. The proper unit of measurement is MMBF. For example, for Alt 2 the harvest level in the first decade is 2,335 MMBF—not 2,335 MBF.

PL-13 S-19 Error in Table S-6: For decade 12, for Alt 1, the harvest level is 1,350 MMBF (see Table 3.9-6b).

[1] McKinley, M. 1997. Large woody debris source distances for western Washington cascade streams. An undergraduate research project done in cooperation with the Washington Department of Natural Resources and the Campbell Group, Inc. College of Forest Resources, University of Washington, Seattle, WA

[2] USDA. 1998. Proceedings of the Conference on Coastal Watersheds: The Casper Creek Story. USDA Forest Service, Pacific Southwest Research Station, General Technical Report PSW-GTR-168

[3] Pacific Watershed Associates. 1998. Sediment source investigation for the lower Eel River, prepared for the United States Environmental Protection Agency, Humboldt County, CA.

[4] Pacific Watershed Associates. 1998. Sediment source investigation and sediment reduction plan for the Bear Creek Watershed, Humboldt County, California, prepared for the Pacific Lumber Company, Scotia, CA

[5] Pacific Watershed Associates. 1998. Sediment source investigation and sediment reduction plan for the North Fork Elk River Watershed, Humboldt County, California, prepared for the Pacific Lumber Company, Scotia, CA

PL-14	S-19	Error in Table S-6: The average harvest per decade for Alt 3 was calculated incorrectly in Table 3.9-6h and incorrectly used in Table S-6. The correct number is 933 MMBF.
PL-15	S-19	Many of the percentages in Table S-6 are incorrect due to all the other errors listed above.
PL-16	S-22	The correct term is Timberland Production Zone or TPZ (see 14 CCR 895).
PL-17	S-25	Much more is known about the potential impacts of herbicides on covered species than revealed in the EIS/EIR. Please review Dr. Dean Thompson's assessment discussed below and submitted as part of this letter as Attachment #III-2. Dr. Thompson does not come to the same conclusion as the draft EIS/EIR authors.
	Chapter 2:	
PL-18	2-2	First paragraph, missing open parentheses in 4th sentence.
PL-19	2-3	Generally, storm-proofing is hyphenated; make this change globally.
PL-20	2-4	End of first paragraph and start of second paragraph are redundant.
PL-21	2-10	Maps say Coppermill Creek. Should be Cooper Mill. Fix page 2-11 also.
PL-22	2-17	Figure 2.5-4: Headwaters Reserve area is not cross-hatched in legend.
PL-23	2-22	Table 2.5-1: footnote 2/ acreage should be referenced.
PL-24	2-25	5th paragraph: 3rd sentence: should be in stream.
PL-25	2-27	1st paragraph: 2nd sentence: should be activities.
PL-26	2-30	In Table 2.5-3a you incorrectly indicate that under the default prescriptions a "size distribution is required" for bands 2 and 3. However, under the default prescriptions the size distribution, as under the interims, is only a target. Both the interim and default allow for substitution from larger size classes. The interim measures further allow substitution from smaller size classes only if substitution by larger size classes is not possible. Please correct this error in the EIS. This error is apparently repeated in Table 2.5-3b.
PL-27	2-42	Table 2.6-1: MMCA total acreage with/without Owl Creek, Alts 2 and 2A, acreage figs should be reversed.
PL-28	2-43	Table 2.6-1: Northern Spotted Owl Suitable Nesting Habitat Harvested-Net Change Alt 1: footnotes do not apply. Riparian Management Zones on PALCO ownership, Alt 1, shouldn't there be a range, or a footnote stating that most conservative option is shown?
PL-29	2-46	Table 2.6-2, page 1 of 8: Again, the interim prescriptions are incorrectly differentiated from the defaults on the basis of a "required size distribution". See comments to Page 2-30 above, the same applies here. In addition, as noted in the Part IV, Section 5 of the HCP (Footnotes for Incremental Benefits Analysis-Evaluation of Canopy Temperature) an audit of canopy levels under current Forest Practice Rules demonstrated that canopy levels of more than 70 percent result from current practices. This study is appended to Part IV, Section 5, so you can review it for

yourself. Given that Forest Practice Rules meet agency targets for shade/canopy, and that PL's interim measures include even more restrictive tree retention levels (e.g., no harvest and late seral buffers) how do you justify and explain your finding that the interim measures could create "additional risk of potential effects on stream shade in localized areas."? Similarly, what basis are you using to conclude that the "absence of marked trees" has any measurable effect on canopy levels? If the no harvest buffers included within the interim measures as part of the requirements of AB 1986 are factored in, the interim measures should be more protective of shade/canopy than are the default measures.

PL-30 2-49 Table 2.6-2, page 4 of 8: Again, the interim measures are criticized for potential impacts to LWD and shade/canopy. Given our comments on Table S-3, Page S-15, Page 2-30, and Table 2.6-2, page 1 of 8 presented above we believe that this criticism is inaccurate. Please indicate what scientific studies or analysis you used to conclude that the interim measures could lead to localized impacts on LWD and shade. In addition, please explain why the no harvest buffers included within the interim measures as required by AB 1986 do not make the interim measures more protective of LWD and shade than the defaults.

PL-31 2-54 2.6.2.1 Alt 1, 1st paragraph: delete "areas that are sometimes called lesser cathedrals or", substitute "smaller stands such as the"...

PL-32 2-55 2.6.2.2. Alt 2, 1st paragraph, 2nd sentence: delete "the only". Add after 2nd sentence a new start to 3rd sentence: "Although the Reserve would contain old growth Douglas-fir trees as part of the species mix,"

PL-33 2-57 3rd paragraph: 4th sentence: delete "degraded", substitute "previously converted". Same comment page 2-59, 3rd paragraph.

PL-34 2-58 Douglas-fir should be hyphenated. Make this change globally.

PL-35 2-63 The discussion of the AB 1986 conditions on this page discusses only the benefits of additional property acquisitions and murrelet restrictions. The aquatic mitigations required by AB 1986 (e.g., no harvest buffers) need to be listed here as well. The reader should also be directed to the discussion of the benefits of these mitigations that are presented later in the EIS.

PL-36 2-64 4th paragraph, 1st sentence, should be "the PALCO...". 8th sentence: "term" is duplicated.

PL-37 2-67 The EIS notes that benefits of the road programs in the EIS on "influxes of coarse sediment to streams." The EIS at this point should also note that the road programs contained within PALCO's EIS will reduce delivery of fine sediment from surface erosion of roads as well. Later on that page the EIS notes that the "winter road management prescriptions could result in adverse impacts..." That should be corrected to read the "winter road construction prescriptions..."

PL-38 2-68 3rd paragraph: 5th sentence: delete "of", add "MMCA" after Creek. Same comment page 2-72.

PL-39 2-68 What scientific evidence do you have to conclude that "Increased frequency of flooding in some streams located below PALCO lands is related to coarse sediment influxes that have aggraded channels"? PALCO is aware of no such scientific evidence. Indeed, demonstrating that flooding levels have increased would require: 1) a long term record of stream discharge to demonstrate that more frequent flooding, if it had really occurred, was not due to an increase in high discharge events, and 2) some record of the maximum stage level of individual flood events to document that a given discharge level was really leading to a higher stage elevation currently compared to during historic periods. Is this type of information available? If not, then again on what do you base your comment? We would also refer you to the study conducted by Hugh Scanlon and Pete Cafferata of the California Department of Forestry on bed elevations in the Freshwater basin. We have attached this study for your review (see Attachment I-1). Contrary to claims of some residents in Freshwater, Mr. Scanlon and Mr. Cafferata found that the bed elevation in two transects of Freshwater Creek downstream of PALCO's land had actually lowered or was unchanged relative to levels documented in the past. This demonstrates the risk of using anecdotal accounts of residents to conclude that PALCO's operations have significantly affected stream bed elevation or composition in areas downstream of our ownership. If your conclusion of increased flooding is based all or in part on resident's comments, the EIS should acknowledge the low scientific certainty associated with such comments.

PL-40 2-74 Statement that "all nesting habitat for the northern spotted owl habitat" would be protected is not true. Not all residual habitat is nesting habitat for spotted owls. Selective harvest will reduce suitability of second growth nesting habitat, and will produce less dusky-footed woodrat habitat than clear cuts.

PL-41 2-78 2.9.3.5, Northern spotted owl, delete "to" before "be".

Chapter 3.4:

PL-42 3.4-11 In the discussion of natural sediment production (see the top of the second column) you mention the Mendocino Triple Junction. Are you suggesting that this is the cause for natural sedimentation? If so, you only have part of the "equation". You may want to elaborate further, possibly talking about other natural events such as wildfire and other causes such as natural erosional processes.

PL-43 3.4-15 PACLO has previously provided data and studies to the agencies documenting that they (the agencies) have calculated incorrectly calculated MWAT criterion. The essence of the error involves the selection of an artificially low and inappropriate OT, or optimum temperature. We suggest that Foster-Wheeler technical staff review the agencies' calculation of MWAT to determine for itself whether the calculation of MWAT is incorrect. This concern is relevant because this section of the EIS references specific temperature data for streams on and in the vicinity of PALCO's ownership and makes several conclusions about where temperatures are a problem, and how canopy is a "critical indicator" of stream temperatures. However, the EIS fails to note that one stream that exceeded MWAT values is an old growth dominated stream in Humboldt Redwoods State Park (Canoe Creek). Volume II, Part F of the HCP, under the Stream Bed Survey data for the Eel WAA, indicates that the Canoe Creek Watershed is 62 percent old growth (i.e., only 38 percent has any harvest history). In addition, no harvest has occurred in this watershed in over 40 years. The EIS should disclose that MWAT values were exceeded in an old growth dominated stream. Further, if the agencies MWAT values are exceeded in an old growth dominated stream the EIS should present evidence or analyses that demonstrate why the agencies' MWAT value is not inappropriate for the region where PALCO's lands are located. Finally, the EIS should discuss why its conclusion that canopy is a "critical indicator" of stream temperature is valid even though MWAT exceedences were noted in an old growth dominated stream, as well as in the North Fork Elk River, where canopy exceeded 93 percent.

PL-44 3.4-19 Inclusion of fecal coliform data in the EIS is welcome. However, the EIS should note that these fecal coliform levels were measured below PALCO's ownership, in an area dominated by residential development utilizing septic systems. As written, the EIS could incorrectly suggest to the reader that fecal coliform levels are necessarily associated with PALCO's activities.

PL-45 3.4-19 Only about a third of the Van Duzen is within PALCO ownership. Or are you stating that for most of the Van Duzen with PALCO lands has a floodplain?

PL-46 3.4-19 In your discussion of aggradation, perhaps it should be "...during the last 40 years..." The '64 flood obviously was over 30 years ago.

PL-47 3.4-20 Your discussion on rain on snow (ROS) is not particularly relevant to PALCO's ownership. PALCO lands do not have significant snowfall or accumulation let alone ROS events. There is no argument that ROS events within the upper portions of watersheds (not on PALCO timberland) played important roles. Perhaps you can explain this within a watershed framework with the location of PALCO lands and the cumulative effects?

PL-48 3.4-20 In your discussion on what is a hydrologically immature forest we have a concern that this may not be the case for the forest ecology here. This is not a hemlock, Pacific Silver Fir; Douglas-fir dominated forest that you find in western Washington – within which these hydrologic studies were completed. We understand and appreciate the need to evaluate this but do we really know that this is the proper method for Redwood forests?

PL-49 3.4-21 In the first paragraph you have answered some of our earlier questions here. Is there someway to put some of this at the front of the section in a topic sentence or two?

PL-50 3.4-21 In the second paragraph you discuss the mixed results of road drainage studies. Thank you for this acknowledgement! You obviously understand and appreciate the statistics or lack thereof in her study. This was an important study and can show some trends but the jury is still out which Ms. Wemple fully acknowledges. Please emphasize this point a bit more than you have here – possibly up front somewhere?

PL-51 3.4-21 Top of the page, in the first column you are continuing your discussion on hydrologically immature forests. Redwood root systems appear to have a higher vitality than other conifers after harvest as observed in the number of young trees sprouting from the root systems. Therefore, we believe that your sentence here, although appropriate for most coniferous forest, may not be "true" for redwoods.

PL-52 3.4-21 Midway through the second paragraph you talk about the mix results of the studies. Yes, but his research was in western Oregon and western Washington where the forest ecology is different. What we need is more research, similar to his, but within redwood dominated forests. Perhaps we need to have this stated a little bit more in this manuscript?

PL-53 3.4-21 In the second column you discuss the possible effect of clearcutting to reducing roots and the possible consequences. Are these germane to redwood forests where we have sprouting root systems?

PL-54 3.4-21 Yes, water yield has been shown to increase in harvested areas in the Pacific Northwest, but in redwood ecosystems we have the fog drip phenomena in which an increase in available water for infiltration is available by the collection of fog on redwood needles. By clearcutting the redwoods there is a decrease in available water. This is explained in several texts (e.g., page 34 of Brooks, K.N., Ffolliott, P.F., Gregersen, H.M., and Thames, J.L., 1991, Hydrology and the Management of Watersheds: Iowa State University Press, Ames.). You should probably at least acknowledge this phenomena in your comments here.

PL-55 3.4-32 Within your discussion of peak flows you mention Class I and II. Please make sure that the reader understands what these mean and which system they belong to.

PL-56 3.4-35 In the second paragraph you discuss an assumption of road width being 30-feet. Is this the average width of the road prism or the traveled way? We are assuming that it is the road prism width, if it isn't you need to adjust the 30 down to 16-feet (standard traveled way width for low volume roads) and correct your calculations.

PL-57 3.4-35 At the bottom of the second paragraph you discuss zero- or first-order basins. You may want to define what a zero- or first-order basin is.

PL-58 3.4-59 This page proposes an agency sponsored mitigation for wet weather road use. Why is this mitigation included in the EIS? At a minimum the EIS is inconsistent in offering this mitigation given that: 1) on page 3.4-58 the EIS concludes that "Except for the high risk of winter road construction increasing discharge of management-related sediment to streams no significant effects are expected. Therefore, no additional mitigation is required except for winter road construction"; and 2) page 3.4-47 concludes that the risk of winter roads use "...has been minimized to a level of less than significant because the HCP requires that road use activities cease when activities result in a visible increase in turbidity." The FEIS should not include this agency mitigation or else scientific data or analyses should be presented showing why the HCP's winter roads use restrictions would not be sufficient to protect aquatic resources.

Chapter 3.5

PL-59 3.5-5 In your quote from Bedrossian you state ... and this can influence slope stability. We suspect that what Trinda was emphasizing is that it may influence slope stability.

PL-60 3.5-6 Land use activities such as vegetation removal, road building, and landing construction do not always result in "destabilization." We know that this wasn't your intention but some readers can misconstrue your statement to make it sound that all harvest and road construction activities cause destabilization. Perhaps you can rewrite this sentence?

PL-61 3.5-8 Under the topic paragraph for section 3.5.2.2. Maybe we are missing something here, but the logic in this statement escapes us and the content is similar to our comment above. How do management activities increase a site's susceptibility to seismically induced mass wasting (coseismic landsliding)? Perhaps you might be able to make your point by stating that seismic accelerations may influence slope instability (increases in driving forces), but to take your statement to the management level doesn't make sense. Again, not all roads, timber harvests, and landings produce unstable conditions. Perhaps you can rewrite this sentence?

Chapter 3.6:

PL-62 3.6-1 The Hookton Formation is not on the geology map (Figure 3.5-1).

PL-63 3.6-1 Sixth paragraph. We understand what you are trying to say here but if you look at this sentence with a "jaundiced eye" it reads as if the Yager Formation has debris slides and flows that form the convex slopes. What forms the convex slopes? Is it the Yager Formation or the debris slides and flows originating from the Yager Formation?

PL-64 3.6-2 First paragraph under Soil Development Factors. Please substitute the word "validation" (or another synonym) for "ground truthing".

PL-65 3.6-15 First paragraph under Roads. Actually, a vast majority of the data for the Cederholm and Reid study was from the Clearwater Watershed, not for the whole Peninsula. There remains some dispute to the extrapolation of data for the Peninsula.

PL-66 3.6-15 First paragraph under Roads. Discussion on improved road standards. This is another problem of Cederholm's and Reid's work. Not all roads were constructed with the same construction methods or materials. A case in point was the work done on "marginal aggregates" by Olympic NF Geotechnical Section, the CTI research completed by the Intermountain Research Station and the ONF Geotech. Section, and roads with low shear strength soils (Koler, T.E., 1995, A soil identification method for potential road construction problem sites on the Olympic Peninsula, Washington: Environmental and Engineering Geoscience Journal, Vol. I, No. 2, pp. 129-137.). In other words, there is more information from other studies that don't agree with Cederholm's and Reid's work. We suggest that you get a hold of Dr. Foltz at the Rocky Mountain Research Station (208 883-2312) and get more information on this topic – or at the very least additional citations than Cederholm and Reid.

PL-67 3.6-17 Deep-seated landslides. This is in conflict with other studies that show that DSLS are the most highly productive areas for conifer forests (e.g., Amaranthus, M.P., Rice, R.M., Barr, N.R., and Ziemer, R.R. 1985. Logging and forest roads related to increased debris slides in southwestern Oregon; Journal of Forestry 83(4)229-233; Burroughs, E.R. 1985a. Landslide hazard rating for the Oregon Coast Range. In: Proceedings of watershed management in the eighties. American Society of Civil Engineers. pp. 132-1369; Burroughs, E.R. 1985b. Survey of slope stability problems on forest lands in the west. USDA Forest Service Pacific NW Forest and Range Experimental Station General Technical Report No. 180, pp. 5-16). Maybe what Dr. Kelsey observed was a dirth of forest regeneration on DSLS associated with soils developed from the Franciscan Formation –which is a possibility. By definition DSLS cover large areas, in my experience they average a few hundred acres in size, and if your statement is true there would be huge areas of treeless areas on PALCO timberland.

PL-68 3.6-17 Mid-paragraph in Deep-seated Landslides – discussion on intergranular friction. Actually it doesn't decrease "intergranular friction" but instead decreases the normal stress as defined by normal effective stress: $\sigma' = \sigma - u$ where u is the uplift pore-water pressure.

PL-69 3.6-17 Same paragraph, discussion on lateral support. This depends on what your definition of lateral support is. How does lateral support decrease when you remove or modify the toe of a landslide?

PL-70 3.6-17 Bottom of page, second column – discussion on Van Duzen watershed. This is doubtful because of several factors. The geology is not consistent across the timberland with what is occurring in the Van Duzen. The soil mechanics are not uniform as you suggest. And the ground water regime (an important factor for DSLS) is highly variable from one tributary to another let alone from one watershed to another.

PL-71 3.6-19 First paragraph. You have similar geomorphic features and yet you appear to have just contradicted yourself as stated above. Please reconsider what you have said in this and the previous paragraph.

PL-72 3.6-19 Second paragraph. There is a more recent version of Varnes classification (Cruden, D.M., and Varnes, D.J. 1996. Landslide types and processes. In:

Turner, A.K., and Schuster, R.L., (editors). Landslides Investigation and Mitigation. National Research Council Transportation Research Board Special Report 247; pp. 36-75.)

PL-73 3.6-19 Third paragraph – discussion on decrease of evapotranspiration increasing potential slope instability. Actually it doesn't for deep-seated landslides but probably does for shallow landslides. From Mr. Koler's research, it was found that average actual evapotransporation (using the Penman-Montieth equation that accounts for stomatal activity of conifers) is along the order of several tenths of a mm/day to maybe a few mm/day, whereas the kinematic wave that responds in the groundwater regime to heavy rainfall has a crest of several cm. Therefore, one can see dimensionally that ET is insignificant in DSLS. In other words, it is the kinematic wave that develops in response to a long wet season and propagates down the slide mass decreasing the normal stress.

PL-74 3.6-20 First paragraph on Surface Erosion. A third source is the storage within the floodplain that is several orders of magnitude higher than the first two sources.

PL-75 3.6-20 First paragraph. You don't have any citations from the WEPP research at the Rocky Mountain Research Station. When you contact Dr. Foltz, he can give you some copies. This is important because their work is showing some different results than the work that you have cited.

PL-76 3.6-20 End of first paragraph – discussion on overland flow. This only occurs if you have extremely shallow soils or the soils have been heavily compacted. Infiltration rates for forest soils range from 70mm/hr to over 200mm/hr and even in the rainy redwood coastal area you rarely get rainfall intensity-duration that is higher than the infiltration rate.

PL-77 3.6-20 Last paragraph. Compaction can certainly occur but you must realize that the compaction creates only a "skin" overlying soil that retains a high infiltration rate. This skin is frequently disturbed so that if overland flow occurs it doesn't travel far before being intercepted by the perturbation. Road fills are compacted in "lifts" of a few inches thick before more fill is placed for compaction. In this case the entire soil column in compacted – something that doesn't happen in logging operations unless the soil is very thin.

PL-78 3.6-21 Maybe we missed it but it appears that your citation of Reid et al. is missing in your references section. Also, how about the citations of Marron et al.; Best et al.; and Megahan and Kidd that you cite in the first paragraph under surface erosion on page 20, and Wilson et al. on page 21?

PL-79 3.6-21 Comment on tractor-yarded schist slopes. There is no schist on PALCO timberland except for some knockers in the Franciscan, so how is this pertinent?

PL-80 3.6-21 Last paragraph on page 21. Not every reader will know what a low-order drainage is. Please describe it.

PL-81	3.6-21	Last paragraph. Comments on the Redwood Creek studies. These are very important studies but they are different from other work in which rainstorms were simulated and data were collected (WEPP work by Rocky Mountain Research Station) for quantifying amounts of sediment produced and delivered. Weaver's and Kelsey's work is excellent but it was based on observations only.
	Chapter 3.9:	
PL-82	3.9-1	WAA is watershed assessment area.
PL-83	3.9-16	The discussion about vertebrate species and their "dependence" on old growth needs to be rewritten to reflect local conditions. For example, in the Plan Area, northern spotted owls are not dependent on an old-growth canopy system for habitat.
PL-84	3.9-17	Late seral forests contain 24" trees. The average DBH is not 24" or greater. Strike the words "that average."
PL-85	3.9-18	This is a strange sentence: "Forests provide society with sustained timber harvests where ***old-growth productivity can be maintained*** due to favorable environmental factors and management polices permit." By managing forests, PALCO plans on dramatically increasing the productivity of its timberlands.
PL-86	3.9-21	THP is *timber harvesting plan*. It is not timber harvest plan.
PL-87	3.9-36	There are at least two sentences on this page that don't make sense: "Therefore, this alternative would be substantially below LTSY (approximately 60 percent). This would result in a less than significant loss of production." Please add additional language to explain this apparent contradiction. How can harvest levels be 60 percent below LTSY and at the same time not be a significant loss of production?
PL-88	3.9-37	In Table 3.9-6a the average harvest per decade is not correct. It is 1,299,084 MBFN.
PL-89	3.9-38	Table 3.9-6c contains errors. First of all, the average harvest per decade has not been calculated correctly. Secondly, note that it is identical to Table 3.9-6d. The entries in Table 3.9-6c should be smaller than the entries in Table 3.9-6d.
PL-90	3.9-39	In Table 3.9-6e the average harvest per decade is not correct. Note that it is larger than the average given in Table 3.9-6f.
PL-91	3.9-40	Something is wrong with Table 3.9-6g and/or Table 3.9-6h. The harvest levels should differ.
PL-92	3.9-41	Both tables (Table 3.9-6i and Table 3.9-6j) contain errors. The average harvest per decade figures are not calculated correctly.
	Chapter 3.10:	
PL-93	3.10-7	Mountain quail genus is "Oreortyx".
PL-94	3.10-8	Isn't osprey CFP?
PL-95	3.10-11	Great egret is CFP.

PL-96	3.10-12	Snowy egret is CFP.
PL-97	3.10-18	2nd paragraph, last sentence, are six California fully protected species on List A.
PL-98	3.10-19	3rd paragraph: 1st sentence, substitute "were" for "was". Delete 6th paragraph, it is covered in list above it.
PL-99	3.10-28	Re: Patch Size, how did Harris (1984) "determine", or was it theory only?
PL-100	3.10-35	1st paragraph, delete "Algumorda", substitute "Algamorda". 3rd para, delete "caurinus", substitute "caurina". 4th paragraph, 2nd sentence, there is much info on benthic macroinverts via the stream monitoring program. 5th paragraph, what is (CWWR, 1996)?
PL-101	3.10-38	The northern red-legged frog discussion does not address the species "intergrade" situation.
PL-102	3.10-42	Population Status, 2nd paragraph, next to last sentence discusses low productivity in 1996, but does not address productivity in 1997.
PL-103	3.10-45	Project Area: Refs should be to Table 3.10-5, which state 17,584 acres of uncut and residual.
PL-104	3.10-47	End of first paragraph, add: "However, a very high percentage of what appears to be potentially suitable habitat has been surveyed, and is accurately assessed as to occupancy in this analysis. What remains is the lowest density, most scattered, and therefore the least likely to contain any murrelets (Appendix N)."
PL-105	3.10-51	end of 3rd paragraph: There is no discussion of CDFG protection measures, i.e., 1,000' buffers.
PL-106	3.10-60	Little Willow Flycatcher, 2nd paragraph, delete "which" from 1st sentence. 3rd sentence, insert "and" before "a federal species". Delete "and is on PALCO's List B". This species is not on List B. Apparently a first nest was discovered in Humboldt County in 1998 (Check with John Hunter FWS, Arcata).
PL-107	3.10-68	Under Northern Goshawk, statement about DFG Code 3503.5 is not present. Should check for all Falconiformes and Strigiformes.
PL-108	3.10-75	1st paragraph, 2nd sentence, should read, "This species nests in hollow trees and snags, in cavities excavated by Pileated woodpeckers, and also in chimneys…"
PL-109	3.10-80	1st paragraph, 4th sentence, insert "the" before Salton Sea. 3rd paragraph, 2nd sentence, substitute "be" for "by".
PL-110	3.10-104	2nd sentence regarding "Class III wetlands" and wildlife, if aquatic life is present, any wetland or watercourse would be a Class II.
PL-111	3.10-105	Snag and Downed-Log Habitat, 1st sentence, delete "eliminated", insert "reduced the number of available".
PL-112	3.10-115	end of 1st paragraph, ridge between Yager and Van Duzen will not be early seral in the long term.
PL-113	3.10-117	Mitigation for Invertebrates, 1st paragraph, should be "mitigate", not "mitigates".

PL-114	3.10-125	2nd paragraph, 4th sentence, after (7,800 acres) insert "consisting predominantly of low density, low quality residual...".
PL-115	3.10-126	2nd paragraph, buffers adjacent to public preserves are late seral prescription, not "no harvest".
PL-116	3.10-130	Northern spotted owl, uses generally old growth forest on federal lands which are primarily Douglas-fir, on managed lands uses primarily second growth redwood dominated forest.
PL-117	3.10-133	4th paragraph, 7th sentence, delete "nesting pairs", substitute "sites containing pairs or single owls". There should be discussion of 75% and 67% thresholds.
PL-118	3.10-161	(and throughout document) when discussing old growth redwood and residual redwood timber types, it would be more appropriate to refer to them as old growth or residual redwood and Douglas-fir.

Chapter 3.11:

PL-119	3.11-9	In the Alternative 1 discussion, HCP prescriptions are addressed at two different points. It seems highly inappropriate to discuss the benefits of the HCP in the No Action/No Project Alternative. These same benefits were excluded from the Proposed Action/Proposed Project Alternative where they should have been acknowledged.

Chapter 3.12:

PL-120	3.12-19	We believe that the second full sentence discussing the likelihood of PALCO purchasing additional timber to maintain sawmill employment is inaccurate. "Section 3.13 speculates that this purchase is unlikely for Alternatives 1, 3, and 4 and quite likely for Alternatives 2 and 2a." It is our understanding that under Alternatives 3 and 4 we are most likely to purchase additional timber and under Alternative 2/2a we are least likely to do so.

Chapter 3.14:

PL-121	3.14-1	The herbicide assessment should be focused on the methods used by PALCO. Discussions and potential effects based on aerial application methods are not relevant.
PL-122	3.14-2	The activity period for pre-emergent herbicides is not necessarily "throughout the rest of the growing season." Please consult a more knowledgeable source of information.
PL-123	3.14-6	PALCO has three major herbicide programs. The first is a pre-emergent program to control weed species before they become a problem. The second is a "rehabilitation" program and involves basal bark applications or "hack and squirt" methods to kill established hardwood and brush species. The third program targets weed species adjacent to roads (e.g., pampas grass). This detail is provided because the EIS/EIR authors got confused and referred to the "rehabilitation" program as a "reforestation"

		program repeatedly on pages 3.14-6, 3.14-9, 3.14-10, and elsewhere. Please fix these errors.
PL-124	3.14-11	It is suggested that more research is needed on the cumulative effects of forest herbicides and then a 1993 reference is provided (i.e., Neary et al. 1993). Well, a lot has been learned since 1993. Please consult more knowledgeable sources of information. Perhaps start with Dr. Dean Thompson's assessment of PALCO's herbicide program (see Attachment #III-2).
PL-125	3.14-19	There are contradictory statements on this page. In one paragraph, it is concluded that "no cumulative environmental effects are expected from the proposed use of herbicides within the Project Area." Two sentences later a different conclusion is reached: "the cumulative effects of herbicide use over the length of the permit period may possibly result in significant effects." Our assessment ,and that of Dr. Thompson (Attachment #III-2), is that the former determination is the correct one, and the PALCO herbicide application, management and monitoring program does not pose potentially significant adverse impacts, direct or cumulative.

Appendix M:

Table M-2: General Comment, the measures listed in the PALCO SYP/HCP column are from previous drafts of the HCP, and need to be updated in accordance with Volume IV of the Public Review Draft of the HCP. There are many errors in this table, and the entire table needs to be edited for accuracy. For example, under marbled murrelet measures, the breeding season has been changed to March 24 to September 15. Should be MMCA, not MCA. Under the northern spotted owl, the nest site protection measures have not been updated to reflect changes made. There is no discussion of the 16 inactive sites that may be taken. Bald and Golden eagle protection measures do not reflect changes in Public Review Draft.

PL-126	Table M-2	Northern Goshawk, All THPs surveyed in Bear-Mattole WAA.
PL-127	Table M-2	Great Blue Heron, etc., only limited harvest is allowed in buffer.
PL-128	Table M-2	Black-shouldered kite, etc., should be "survey 10% of THP acreage".
PL-129	Table M-2	Footnotes section: Footnote 1/ is Spotted Owl Resource Plan, not Management Plan (SORP) not (SOMP). Feet is spelled wrong. Footnote 2/ should read "Conservation Strategy".
PL-130	Table M-5	"Total Unsuitable Nesting Habitat" is a misnomer. Many pairs are currently nesting in low quality, roosting, and some in foraging habitat. Consider changing title to "Other Habitats", and list non-habitat seperately.

Appendix N, Part 1:

PL-131	Page 1	2nd para, 1st sentence, insert "marbled murrelet" before "habitat". 2nd sentence, it is incorrect to say that "few data" are available. Several years of data were used in the RBV approach.
PL-132	Page 2	Douglas-fir is hyphenated. Make this change globally.
PL-133	Page 3	old growth may be hyphenated or not, but should be consistent (make such changes globally). 3rd paragraph, 5th sentence, "circling over" is not generally associated with the sub-canopy behaviors, although considered an occupied behavior in the HCP analysis. Delete "foliage", insert "canopy level".
PL-134	Page 4	3rd paragraph, 1st sentence, rather than "occupied behaviors" should it be detections in general? 4th paragraph, There have been no detections of marbled murrelets in the Bear-Mattole WAA. 5th paragraph, TRA has deleted references to "not adequately surveyed" and inserted "low survey intensity".
PL-135	Page 6	1st paragraph, murrelet eggshell fragments have been found in the Nanning Creek stand, in a residual area.
PL-136	Page 8	last paragraph, Page 9, 1st para, the discussion of "several small stands", and "large uncut stands" is confusing.
PL-137	Page 9	last paragraph, MMCA, not MCA.
PL-138	Page 10	2nd paragraph, 4th sentence, Re: the "Below Road 7 and 9 "stands, occupied behaviors have been detected in the areas within the MMCA, not in the areas available for harvest. Also, surveys of the area have been fairly extensive.
PL-139	Page 23	Need complete reference for Marzluff.
PL-140	Page 24	USFWS ref, *Brachyramphus marmoratus* should be italicized.
PL-141	Page 27	Table N.1-3. There should be footnotes to explain the 1320 and 300 rows.

Appendix N, Part 2:

PL-142	Page 4	3rd paragraph, 2nd sentence, Survey data is collected from <u>April</u> through August. Last sentence, insert "for two years" before "with", in two places in sentence.
PL-143	Table 1.A	Page 2, in footnotes an "R" is missing from REDOGW1.

II. Independent Assessments or Analyses By Consultants or Experts Responsive To Public Comments, Concerns, or Expressions of Need For Further Analysis in the Draft EIR/EIS

ISSUE II-1

THE FINAL EIR/EIS SHOULD NOT BE UNDULY INFLUENCED BY DR. REID'S BIASED REVIEW OF THE PWA BEAR CREEK SEDIMENT SOURCE REPORT

Dr. Bill Weaver of PWA has drafted a critical response to Dr. Leslie Reid's review of the Bear Creek sediment source report. His response is contained in memorandum from Dr. William Weaver to Dr. Daniel Opalach, PALCO's Timberland Manager, and is submitted as part of this letter as Attachment II-1.

ISSUE II-2

THE FINAL EIR/EIS SHOULD CONSIDER THE IMPACT OF HERBICIDE USE ON COVERED SPECIES TO BE CONSIDERED MITIGATED TO INSIGNIFICANCE

Dr. Dean Thompson has provided PALCO with a detailed risk assessment of herbicide use on PALCO's timberlands. Dr. Thompson concludes that the potential effects of PALCO's herbicide program on most species of concern were considered to be mitigated to insignificance. Dr. Thompson proposed a monitoring program to provide additional information pertaining to the minimal risks on red-legged frogs and northwestern pond turtles. Dr. Thompson's report is submitted as part of this letter as Attachment II-2.

III. Information, Data or Analysis from PALCO Responsive To Public or Agency Comments Noted Through the Draft EIR/EIS Review Process

ISSUE III-1

THE FINAL EIR/EIS SHOULD NOT BE UNDULY INFLUENCED BY DR. REID'S BIASED REVIEW OF THE PWA BEAR CREEK SEDIMENT SOURCE REPORT

SCIENTIFIC RESEARCH VS. TECHNICAL REPORT

Dr. Reid completed a thorough review of PWA's Bear Creek sediment source report (PWA 1998) and many of her comments are salient in improving the document. However, the resolution of her review comments were on the order of what one would expect for a scientific research paper. This document is a technical report with the specific purpose to help PALCO management make informed decisions in reducing sediment production and yield to fish-bearing streams. PWA was retained to provide PALCO with a technical report – not a scientific, anonymously peer-reviewed manuscript to be published in a scientific journal. PL-144

DR. REID IGNORES THE IMPORTANCE OF STORMS

Dr. Reid focuses only on timber harvesting and landsliding and ignores the importance of the 1996/1997-storm event. If the data were readily available, it would be critical to assess the timber harvesting that occurred prior to the 1955 and 1964 storm events. Unless such assessments are conducted, how we make the conclusion that Dr. Reid does that, there is a 9.6 increase in landsliding in response to logging? Maybe that 9.6 increase is related to the storm activity, or more likely, a combination of management activities and the storm. PL-145

A REVIEW OF DR. REID'S COMMENTS

The following list refers to the "Specific comments" section of Dr. Reid's review of the PWA's Bear Creek report.

4/5 Yes, logging does influence a decrease in the resisting forces of a hillslope after harvesting. However, it is important to understand its effect is rarely uniform across the slope and in many cases is dimensionally insignificant when compared to other factors such as slope angle and ground water geometry (Hammond et al. 1992). PL-146

4/6 Evapotranspiration is important for shallow soils, but its effect on deep soils is probably insignificant (Koler 1995, Koler 1996). PL-147

4/6 Actually, reduction of foliage in redwood forests decreases amount of water available for infiltration because of the fog drip phenomena (Brooks et al. 1991). PL-148

9/2 This comment goes beyond the scope of this technical report. The purpose of the report was to help PALCO management understand what has occurred in the past and how improvements may be made. There was no modeling provided or even a conceptual design to estimate predicted effects. PL-149

11/2 PALCO understands and appreciates the suggestion for comparing activities and events between two watersheds. However, other factors are more important (e.g., soil and rock mechanics, ground water regime geometry, structural discontinuities in bedrock, etc.) than time-lithologic geology and location of the two watersheds. This observation has been supported by the disparity in the sediment study results for Bear and Jordan Creeks. PL-150

13/3 PALCO concurs with Dr. Reid about the definition of inner gorges. This however is a problem with definitions as provided by the regulating agencies and not with the report. Perhaps there is a way to get CDM&G and CDF to accept Dr. Kelsey's definition? PL-151

14/3 See comments above for 11/2.

15/1 From PALCO's experience with air photo interpretation we are comfortable with what Dr. Montgomery is suggesting. However, there is some recent work coming from the Oregon Department of Forestry (Dent et al. 1997) that suggests just the opposite. From our perspective, it boils down to the expert-level of the analyst and Dr. Weaver is a superior air photo interpreter. PL-152

22/4 Dr. Reid's calculations are incorrect. PALCO did not haul logs out of this watershed every year since 1946. Thus, Dr. Reid's conclusions concerning the importance of surface erosion are erroneous. PL-153

37/2 There is more to the stability story than storms, silviculture practices and road construction. However, Dr. Reid makes several good points about looking at a large area for patterns and similar work in this manner has been done successfully with landtype associations as well as other mapping tools and applications. PALCO disagrees with her last sentence however. Much work has been completed in this area for predicting hillslope instability prior to a particular land use activity (e.g., Hall et al. 1994) PL-154

37/3 This is a "chicken and egg" problem that really cannot be resolved unless we have a good handle on what is really a "natural" event. We can certainly make educated guesses with air photo interpretation and data available from previous surveys and reports. Dr. Reid's point is well taken, but conversely not all landslides are unnatural either – after all mountains do erode over time. Large natural landslide events have been occurring in the Bear Creek watershed for thousands of years before settlement in the 1800s. A quick observational "proof" is the sediment stored in the valley floor. A back-of-an-envelope calculation (12,000 ft long by 300 ft wide by 40 ft deep) indicates that the main stem of Bear Creek has approximately 5.3×10^6 yd^3 in storage and probably much more. Input and output values will fluctuate as the sediment PL-155 is transported but for sake of argument let's assume that they are in equilibrium. Dr. Weaver has estimated that 1,450,000 yd^3 has been yielded in the time of interest in his evaluation – or roughly a quarter of the estimated storage. The implication here is that "large" volumes of sediment are transported through and stored within this fluvial system. Unfortunately, we do not have a frequency for the natural events, but it should be obvious that natural events are a dominant feature and Dr. Weaver's semantics are acceptable when one considers the "bigger picture." PL-155 con.

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Brooks, K.N., Ffolliott, P.F., Gregersen, H.M., and Thames, J.L. 1991. Hydrology and the Management of Watersheds. Iowa State University Press, Ames, Iowa. pp. 34.

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Hall, D.E., Long, M.T., and Remboldt, M.D. 1994. Slope Stability Guide for National Forests in the United States. USDA Forest Service Washington DC Engineering Staff Publication EM-7170-13. 1091 p.

Hammond, C.J., Hall, D.E., Miller, S., Swetik, P. 1992. Level I Stability Analysis (LISA) Documentation for Version 2.0. USDA Forest Service Intermountain Research Station General Technical Report INT-285, Ogden, UT. pp. 32-38.

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Koler, T.E. 1995. Application of Dynamic Ground Water Models in Erosion Analyses. McCammon, B., moderator, Region Six Workshop for Hydrologists, June 1, Portland, OR.

Pacific Watershed Associates. 1998. Sediment Source Investigation and Sediment Reduction Plan for the Bear Creek Watershed, Humboldt County, California. Report prepared for The Pacific Lumber Company, Scotia, California. 42 p.

ISSUE III-2

THE FINAL EIR/EIS SHOULD FIND THAT THE RECENT APPELLATE OPINION IN NATIONAL MINING ASSOCIATION V. ARMY CORPS OF ENGINEERS DOES NOT AFFECT THE DRAFT EIR/EIS REVIEW OF PALCO GRAVEL OPERATIONS

PALCO'S EEL RIVER GRAVEL EXTRACTION OPERATIONS WILL CONTINUE TO BE MONITORED, REVIEWED AND CONDUCTED ON A SUSTAINED YIELD BASIS BASED UPON REVIEW AND RECOMMENDED EXTRACTION LOCATION AND VOLUME, MITIGATION AND MONITORING RECOMMENDED BY CHERT TEAM

BACKGROUND

Extensive description and summary of PALCO's Eel River gravel extraction operations is detailed in the draft HCP at Volume III, part I. Much more detail was provided in pre-review draft materials submitted to the Administrative Record prior to submission of the draft HCP/SYP, including the reclamation plans and detailed aerial photographs, cross-sections and other analyses. Literally thousands of pages of information and documentation was provided to the agencies in these submissions, public record documents on file with various state, local and federal agencies.

Currently, and through 1998, PALCO has provided detailed monitoring and engineered cross-section information on an annual basis to the United States Army Corps of Engineers ("USACE") pursuant to its "Letter of Permission Procedure, Gravel Mining and Excavation Activities Within Humboldt County" ("LOP 96-1"). The purpose of the LOP procedure was to streamline Section 404 of the Clean Water Act authorization for excavation and related work not posing significant adverse individual or cumulative impacts. The USACE LOP 96-1 detailed various terms, conditions and monitoring activities required for consistent evaluation and decision making with the LOP process, and the environmental impacts of the LOP were assessed by USACE in accordance with the National Environmental Policy Act ("NEPA") and Section 7 of the Federal Endangered Species Act ("ESA"), for USACE also initiated a consultation with the National Marine Fisheries Service ("NMFS"). That review process concluded with a finding of no significant impact ("FONSI"), and NMFS issued both (1) a biological opinion ("BO") regarding potential for operations conforming with the LOP to impact salmonid fish species of special concern; (2) an incidental take statement ("ITS") for coho salmon. The LOP 96-1 was approved and became effective on August 19, 1996. The LOP 96-1 has a three year term for review and revision.

MULTI-AGENCY REVIEW: CHERT SCIENTIFIC OVERSIGHT

Throughout Humboldt County, the LOP 96-1 process has been integrated into a multi-agency oversight approach – the most integrated approach to regulatory oversight of the aggregate industry anywhere in California. The process requires that specific mining plans be prepared and submitted in a standard format for review and approval prior to execution. The County of Humboldt Extraction Review Team ("CHERT") independently reviews all such plans and issues a specific recommendation for volume, location, and method of extraction on such plans. In addition, the process requires that specific monitoring data be submitted annually. FWS, NMFS and the CDFG review the recommendations.

In addition to the California Surface Mining and Reclamation Act compliance, County permit procedure, and the LOP requirements integrated into the CHERT review, near stream mining also requires an annual review and permit by CDFG in the form of a streambed alteration agreement, generally known as a 1603 Agreement. This process has been in place for many years. In recent years, requirements for annual monitoring, such as the use of river channel cross sections, have been formulated and standardized to meet environmental needs and to provide continuity and conformance monitoring information needs of all the related agencies. The independent, scientific review process was instrumental in helping to prepare river monitoring guidelines, particularly for cross-sectional survey data.

THE DECISION IN NATIONAL MINING ASSOCIATION V. U.S. ARMY CORPS OF ENGINEERS No. 97-5099 (D.C. Cir. June 19, 1998)

In June of this year, the Federal Court of Appeals for the D.C. Circuit has ruled that USACE and EPA overstepped their authority when they adopted rules to regulate excavation in wetlands. Very briefly, the Clean Water Act of 1972 regulates discharges of pollutants, including "dredged and fill materials," into waters and wetlands. In 1993, the USACE and EPA adopted regulations claiming authority to regulate any activity in waters or wetlands that (1) results in some movement of soil and (2) destroys or degrades the waters or wetlands. The agencies accomplish this by redefining "discharge of dredged materials" to include "any addition, including any redeposit, of dredged material, including excavated material, into waters of the United States which is incidental to any activity, including mechanized land clearing, ditching, channelization, or other excavation."

USACE and EPA contended that the excavation rule provided authority to regulate the "incidental fall back" that generally occurs during excavation and land clearing, including near stream excavation of aggregate. Several trade associations sued the Corps and EPA, and the District Court decided that "the agencies unlawfully exceeded their statutory authority" and enjoined them from applying the excavation rule anywhere in the nation.

The agencies appealed, and the U.S. Court of Appeals of The District of Columbia circuit upheld the District Court decision.

EFFECT OF COURT DECISION ON USACE LOP PROCESS

PL-156

During the public comment period, some have suggested that the LOP process has been undermined or eliminated as a result of the National Mining case. To date, PALCO has received no notice or indication from USACE that it no longer has jurisdiction or will no longer participate in the now fully integrated Federal, State and County review process described above. In fact, PALCO has submitted all of its LOP information for process by the Corps, the County, the State Department of Fish and Game, FWS, NMFS and others for 1998. All of the monitoring and cross-sectional information will still be reviewed by the CHERT team.

In personal conversation with Army Corps of Engineers' counsel, Ms. Merry Goodenough, recently, our counsel was told that the Army Corps may still have jurisdiction, and assert that jurisdiction, based upon the fact that the State Department of Fish and Game often requires land-leveling in near-stream aggregate deposits following excavation. According to Corps counsel, such land-leveling would be considered "fill," bringing the mining activity under the jurisdiction of the USACE.

Furthermore, we have had conversation with the Deputy County Counsel of Humboldt County, who has indicated that the CHERT review process will remain in full force and effect, with the inter-disciplinary scientific review team considering the same information, under the same process, as it has previously, to make the same recommendations regarding allowable extraction levels and monitoring and mitigation measures.

Therefore, in brief, no matter what the impact of the National Mining case may be - and it is as yet uncertain and no announcement as to that effect has yet been made by USACE - the gravel extraction planning, permit, and monitoring requirements as detailed in the draft HCP will remain in effect, unchanged, providing the same information and data to the same agencies for review and consideration.

ISSUE III-3

THE FINAL EIR/EIS SHOULD RECOGNIZE THAT INFORMATION ON "CURRENT" CONDITIONS IN THE HCP IS NECESSARILY DATED

During public hearings on the HCP numerous speakers have claimed that the HCP, and therefore the EIS/EIR, is flawed because information on habitat conditions, fisheries, road PL-157 mileages, estimates of harvested acres, etc. do not reflect current information. PALCO would like to point out that this situation, when present, is usually unavoidable because "current" conditions are changing on a monthly, weekly, or even daily basis. For example, the number of acres of timberland harvested by PALCO changes daily as ongoing harvesting operations proceed. Similarly, many of the descriptions of fisheries and stream habitat are based on surveys conducted in previous years. In many cases no more recently collected information is available, or, if new data have been collected (e.g., during the 1998 field season) they have not yet been computerized, checked for errors, subjected to statistical analysis, etc. PL-157 CON.

Thus, it is true that some conditions on PALCO's ownership may have changed relative to their descriptions in the HCP (e.g., conditions in Bear Creek), but again, this reflects the availability of more current information, and the logistical difficulty of updating the HCP given ever changing conditions on the ground. In preparing the HCP, PALCO's goal was to use the best available information to advise government agencies and the public as to the overall condition of fish and wildlife habitat and management activities on our lands. We believe that we have succeeded in this goal, and that the description of current conditions in the HCP is more than adequate for an evaluation of alternatives in the EIS/EIR.

It is important to note that in the HCP PALCO acknowledges the above described limitations on its writeups of current conditions. For example, page 26 of Volume I acknowledges that the "LTSY [Long Term Sustained Yield] projection in this plan is based on PALCO's inventory as of January 1, 1998." Similarly, Volume III, Part H, Section 1.2 clearly identifies that the datasets used to describe stream and fisheries conditions were based primarily on data collected during or before 1997.

ISSUE III-4

THE FINAL EIR/EIS SHOULD RECOGNIZE THAT FUTURE MANAGEMENT OF PALCO LANDS IN NOT AN "UNKNOWN" PL-158

Another comment made during public hearings on the HCP is that the watershed analysis process proposed by the company makes it impossible to know what future management of PALCO's lands will be. Similarly, some speakers have claimed that watershed analysis can be used by the company to significantly reduce the level of stream/fish protection required under the incidental take permit.

As to the first point, much of the management of PALCO's lands is known with certainty, regardless of the results of watershed analysis. This certainty comes from California's Forest Practice Rules, Assembly Bill 1986, and from a memorandum from the National Marine Fisheries Service (NMFS) dated 3 February 1998. These documents impose nonalterable requirements for terrestrial/aquatic resource protection on PALCO's lands including: PL-158 con.

- All Class I streams must have at least a 30 ft wide "no cut" buffer on either side (required by AB 1986, NMFS). In addition, the results of watershed analysis can be used to require up to a 170 ft wide "no cut" on either side of Class I streams. We put "no cut" in quotes because harvest that enhances riparian function and that is approved by both NMFS and the State of California would be allowed. However, PALCO's experience in implementing this type of buffer on our lands in the past 6 months has demonstrated that in virtually all cases the agencies will require that these buffers be treated as total no cut zones.

- All Class II streams will have at least a 10 ft wide "no cut" buffer on either side (required by AB 1986 and NMFS), and will almost certainly have a 30 ft wide "no cut" buffer on either side.

- Restricted harvest buffers must be present along streams outside of the no harvest buffers (Forest Practice Rules). At a minimum, these buffers will extend out to at least 75 ft (Class Is) or 50 ft (Class IIs) from the active stream channel.

- All timber harvesting, including salvage logging and other management activities that are detrimental to the marbled murrelet is prohibited from all marbled murrelet conservation areas for the life of the incidental take permit (AB 1986).

- Mitigations for road related activities (e.g., construction, hauling, etc.) must be no less protective of species and habitat than those imposed by the February 27, 1997 Pre-Permit Application Agreement in Principle (AB 1986). This means that stormproofing and inspection of roads will continue following the assessment protocols outlined in the HCP. Similarly, PL must utilize wet weather restrictions on road use and construction.

- All California Forest Practice Rules, including regulations governing stream buffers, roads, harvesting operations, winter operations, etc. must be complied with. Thus, Forest Practice Rules are the minimum or "floor" level of protection, with watershed analysis providing only more restrictive or protective measures.

Other aspects of the management of PALCO's lands are not known with the same certainty level as the items above, but nonetheless are extremely likely to be included in any future management prescriptions. This is true because of the overwhelming scientific evidence/support for such mitigations, and/or because such measures have been applied in every approved timberlands HCP in the Pacific Northwest, and/or because watershed analyses using the Washington DNR method regularly or always result in such measures. These extremely likely measures include:

- Restricted entry/harvest buffers along Class I/II streams above those required by Forest Practice Rules. It is virtually certain that watershed analysis will result in the development of such "enhanced" buffers. These enhanced buffers will vary in width and in specific harvest restrictions to reflect site specific conditions, with wider buffers and more restrictive buffers where channel migration zones, unstable slopes, or other conditions require them, and narrower buffers in areas with bedrock canyons, stable slopes, etc. PL-158 con.

- A mass wasting avoidance strategy that limits or prohibits harvest on areas of PALCO ownership that have a high potential for landslides and other mass wasting processes. Current GIS based analyses indicate that at least 30,000 acres of PALCO land (15 percent of the total) have a high, very high or extreme mass wasting potential, and would be subject to a mass wasting avoidance strategy. This figure will likely change, as ongoing scientific studies to determine where landslides occur, and what role timber management has in inducing such landslides, progresses (see for example Sediment Source Inventories of Bear Creek and the North Fork Elk River by Pacific Watershed Associates).

Finally, the HCP contains two fish and wildlife "performance standards" that the company must meet. Thus, the public knows that regardless of the results of watershed analysis, several defined levels of resource protection will be provided by PALCO. These performance standards include:

- Conditions in streams on PALCO's lands will exhibit recovery toward agency defined "properly functioning conditions." These properly functioning conditions include standards for water temperature, fine sediment, mean particle size, canopy cover, large woody debris volume, and pool abundance/size. Thus any watershed analysis must produce mitigations that result in recovery toward these properly functioning conditions.

- Spotted owls on PALCO's lands must persist through time. If spotted owl numbers on PALCO's lands fall below 75 percent of their baseline number, the HCP commits the company to consult with the US Fish and Wildlife Service (USFWS) and California Department of Fish and Game (CDF&G) to determine why owl numbers are falling, and what needs to be done to reverse the decline. If spotted owl numbers fall below 67 percent of their baseline numbers the HCP commits the company to meet with USFWS and CDF&G to develop and implement a "no take" plan for owls.

In summary, the use of watershed analysis does not make future management of PALCO's lands an unknown. Many of the protections that will be provided for fish and wildlife are known with certainty or near certainty. In addition, the outcome of the DNR watershed analysis process in Washington State can be used as a guide as to the types of mitigations that can be reasonably expected from the same process on PALCO's lands. These Washington State watershed analyses have not generally resulted in mitigations and management prescriptions markedly different or less protective that those listed as interim measures in PALCO's HCP as Foster-Wheeler staff should know through their own experience with the DNR watershed analysis process. Instead, the prescriptions are usually very similar to those in PALCO's HCP but with the difference that the mitigation levels are modified for site conditions, being more protective in sensitive sites, and less protective in less sensitive sites. This site specific adaptation of fish and wildlife protections represents the best available approach to managing forest lands in the Pacific Northwest and is therefore likely to make PALCO's HCP more protective overall than the "one size fits all" approach contained in the interim measures. Please re-read that last line, it is the primary reason why the company has even proposed doing watershed analysis as part of its HCP.

PL - 158 con.

As for the second point, that PALCO can utilize watershed analysis to significantly weaken protective measures in the HCP, this is clearly untrue. Under PALCO's draft HCP it was the agencies, not the company, that had a veto over mitigations and prescriptions developed as part of watershed analysis. Accordingly, even if PALCO proposed significantly weaker mitigations than the interim measures, the agencies were completely empowered to reject these modified measures and to impose the even stricter default prescriptions. Since the subsequent passage of AB 1986, the agency role in developing prescriptions from watershed analysis has been significantly enhanced, and the agencies still retain veto power over these prescriptions.

ISSUE III-5

THE HCP CONTROLS MANAGEMENT RELATED SEDIMENT DELIVERY FROM CLASS III WATERCOURSES

PL- 159

Again, several public comments have addressed the concern that the HCP will not control management related sediment delivery from Class III streams to downstream watercourses. This is not true for a variety of reasons:

- Class III streams are small, ephemeral channels with high sediment storage and a limited ability to transport sediment downstream. For example, in one experiment Duncan et al.[6] found that no more than 45 % of the sediment introduced to two ephemeral streams was ultimately transported downstream to the mouth of the tributary they studied.

[6] Duncan, S.H., R.E. Bilby, J.W. Ward, and J.T. Heffner. 1987. Transport of road-surface sediment through ephemeral stream channels. Water Resources Bulletin 23(1): 113-119.

- Sediment delivery from Class III streams typically occurs during periods of heavy precipitation and runoff. However, both Duncan et al.[6] and Megahan[7] found that when sediment was mobilized from ephemeral channels at high flows that downstream channels typically were also experiencing higher discharge and sediment transport rates and that, therefore, these downstream areas had a lower depositional capacity. Thus, a significant proportion of the sediment delivered from Class III streams will be immediately flushed out of the stream system. PL-159 con.

- The HCP includes measures that will limit harvest related ground disturbance and exposed soil along Class III streams. For example, the HCP requires a 25 to 100-foot Equipment Limitation Zone (EEZ) or Equipment Exclusion Zone (EEZ) (depending on slope). These zones will minimize the use of heavy equipment within and adjacent to Class III streams. Where equipment entry in these zones is planned, the access corridors must be mapped in the THP and flagged on the ground. This will allow the agencies to review the proposed equipment usage both in the office and on the ground during the pre-harvest inspection to determine the site specific potential for delivery of sediment to Class IIIs. In addition, there is to be no ignition of fire in the ELZ or the EEZ and all skid trails must be stabilized according to the CFPRs.

- Even where ground disturbance occurs adjacent to or within Class IIIs, disturbed ground on PALCO's ownership typically "greens up" in 1-3 years following harvest. Thus, after harvest even if sediment delivery to Class III streams increases this will be a very short term impact.

- The HCP requires that all down wood both within Class III channels and the adjacent EEZs/ELZs must be retained in place. This provision was specifically requested by the National Marine Fisheries Service to limit sediment movement into Class IIIs and delivery of that sediment into downstream watercourses. At least two studies (Brown[8], Megahan[7]) have confirmed that ground covered with organic debris produces far less sediment than disturbed ground. Other work has confirmed that the sediment storage capacity of a channel, especially an ephemeral channel, is enhanced by the presence of large organic debris (O'Connor and Harr[9], Megahan[7]).

- The HCP's mass wasting avoidance strategy contains default prescriptions that will limit permissible operations in those portions of Class III streams that are susceptible to mass failures (e.g., steep or unstable slopes). The mass wasting avoidance strategy

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[7] Megahan, W.F. 1981. Effects of silvicultural practices on erosion and sedimentation in the interior west- a case for sediment budgeting. Pp. 169-181 *in* Interior West Watershed Management, Proceedings of a Symposium held April 8, 9, and 10, 1980 Spokane, Washington, D.M. Baumgartner, ed. Washington State University Cooperative Extension, Pullman, WA 288 pp.

[8] Brown, B. G. 1983. Forestry and water quality. O.S.U. Book Stores, Inc., Corvallis, Oregon, second edition 142 p.

[9] O'Connor M., and R.D. Harr. 1994. Bedload transport in an Oregon Coast Range stream. Water Resources Bulletin 17(5):886-894.

will also result in site visits by Registered Geologists to many Class III streams to evaluate their potential for sediment delivery to downstream watercourses. These site visits would be used to design site specific prescriptions, if appropriate. PL-159 CON.

- The HCP requires an extensive road maintenance/upgrading program, as well as limits on winter road construction and use. The result of these road related measures is that the amount of both water and sediment delivered to Class III streams from PALCO's road system will be significantly reduced over levels that would occur if the HCP is not approved.

- Retention of a limited harvest or no harvest buffer along Class III streams, as called for by some members of the public, is designed primarily to prevent delivery of sediment from surface erosion surfaces. This is true because the mass wasting avoidance strategy already requires restrictions on road building and harvest for those portions of Class IIIs that are susceptible to mass wasting. But three sediment source inventories of PALCO's lands have demonstrated that mass wasting processes represent, by far, the majority of the sediment delivered to streams (see PWA references cited above). These studies found that surface erosion from all sources constituted less than 10% and perhaps less than 5% of the total sediment delivery to streams. And erosion into Class III streams, in turn, would represent only a fraction of this 5-10 percent. Thus, even if no harvest/limited harvest buffers were required by the HCP, the amount of sediment delivery to streams they would prevent is a miniscule portion of the total sediment budget for PALCO's watersheds. This must be weighed against the enormous financial cost, and operational impairment PALCO would suffer if no harvest/limited harvest buffers were uniformly required along Class III streams.

ISSUE III-6

EPIC HAS ERRONEOUSLY CONCLUDED FROM THE DRAFT EIS/EIR THAT THE FIRST DECADE OF THE PLAN IS ONLY FOUR YEARS LONG PL-160

The harvest schedules contained in the draft SYP/HCP and the draft EIS/EIR were developed by resource professionals at VESTRA Resources, Inc. The procedures were reviewed by Drs. Greg Biging and Larry Davis. The projected harvest, growth, and inventory volumes shown in Tables 3.9-6a to 3.9-6j are shown by 10-year periods. Each and every period is ten years in length. Because of this erroneous interpretation, EPIC has distributed a number of misleading statements to the public via their Web site. For example, *"During this four-year period, over 25% of the company's land will be logged (54,382 acres.)"* Such statements grossly misrepresent the content of the draft SYP/HCP and draft EIS/EIR.

That concludes our comments. Please contact me at 707/764-4117 if you have any questions about this letter or the attachments.

Very Truly Yours,

THE PACIFIC LUMBER COMPANY

Daniel Opalach

Daniel Opalach, Ph.D.
Timberlands Manager
RPF #2459

Encls.

cc: Mr. Thomas M. Herman
Mr. Frank Bacik, Esq.

Attachment I-1

Mr. John Marshall's (CDF) Memorandum to Mr. Dave Ebert (CDF) Concerning the Minor Channel Aggradation of Freshwater Creek

State of California **The Resources Agency**

Memorandum

To: Mr. Dave Ebert
Ranger Unit Chief
Humboldt-Del Norte Ranger Unit

Date: September 11, 1998
R43

Attn: Mr. John Marshall
Resource Manager
Humboldt-Del Norte Ranger Unit

Telephone: ATSS( )453-9455
(916)653-9455

From: **Department of Forestry and Fire Protection**

Subject: 5400 FOREST PRACTICE REGULATION
5410 Forest Practice Act
Freshwater Creek Cross-Section Remeasurement

Background Information

This memorandum reports the results of field work completed on August 19-20, 1998 by Hugh Scanlon and Pete Cafferata, CDF, in the lower Freshwater Creek basin. The purpose of the project was to remeasure three cross-sections established by the U.S. Army Corps of Engineers (ACE) in 1975. The ACE included these cross-sections in their report titled "Flood Plain Information, Freshwater Creek, Humboldt County, California." These cross-sections (along with 6 others that were not included in the report) were established at that time to aid in the determination of the extent of the flood plain defined by a 100-year recurrence interval discharge event. The ACE's goal was to promote proper land use planning and management decisions regarding flood plain utilization in the Freshwater Creek basin.

The cross-sections included in the ACE report are shown in Figures 1 and 2. We were unable to locate bench marks for these cross-sections, so remeasurements here are only approximations of the exact locations surveyed in 1975. We used an engineering level, Philadelphia rod, and 300 foot nylon tape for measuring the cross-sections (equipment provided by Humboldt State University's School of Forestry). Brief descriptions of each site follow, as well as our interpretations of the data collected.

Cross-Section No. 3

Cross-section no. 3 was established by the ACE at river mile 2.74 (river mile was defined by the ACE as miles above Freshwater Corners; Freshwater Slough begins slightly below this point). Based on maps and diagrams included in the ACE (1975) report, we determined that the location of this cross-section was approximately 1000 feet below Howard Heights Bridge at a bend in the channel (see Figure 3). Channel gradient at this location is 0.2 percent (as determined from Figure 4). Abundant willow trees and blackberry thickets made surveying at this location extremely difficult, and we were able to make measurements over about 120 feet, compared to the ACE's cross-section distance of 800 feet. We are relatively confident that we established the cross-section in roughly the same location reported by the ACE, but clearly the remeasurement is only an approximation. Additionally, since actual reference datums were not available for this survey work, we simply assumed that the elevation of high right bank just above the channel where the instrument was set up remained constant and could serve as an approximate reference elevation. Figure 5 shows the comparison of the 1975 cross-section with the 1998 cross-section. Very minimal channel aggradation is shown, with some evidence that the thalweg may have shifted slightly toward the right bank (looking downstream). Since the location for this cross section was at a bend in the channel in a low gradient flood plain, thalweg migration would be expected to occur at this location. Rivers move laterally in a flood plain by erosion of one bank and simultaneous depositions on the other. Throughout the process of lateral movement, the channel generally maintains a similar width and depth (Dunne and Leopold 1978).

Cross-Section No. 5

Cross-section No. 5 was established at river mile 3.74. Based on maps and diagrams included in the ACE report, we determined that this cross section was originally located approximately 600 feet above the confluence of Little Freshwater Creek (Figure 6). Channel gradient at this location is approximately 0.2 percent. The creek at this location has multiple over-flow channels, with sediment terraces spread over several hundred feet in the flood plain. It is obvious that the dominant channel will change over time as sediment routing occurs, and this appears to have been documented with our cross-section remeasurement. In 1975, the deepest channel was located near the right bank (looking downstream), while in 1998 the dominant channel was found further towards the left side of the flood plain. We surveyed about 300 feet at this location, compared to the ACE's survey of 700 feet. Again, since actual reference datums were not available, we assumed that the elevation of a levee on the right bank remained constant and could serve as an approximate reference datum. Based on a comparison of the two cross-sections, it appears that approximately one foot of fine sediment may have accumulated at this low gradient location (Figure 7), although no volume calculations were made to determine mean bed elevation change.

Cross-Section No. 7

Cross-section no. 7 was established by the ACE at river mile 4.70. Based on maps and diagrams included in the ACE report, we determined that the location of the cross-section was approximately 50 feet above the Freshwater Park Bridge (of the three cross-sections, we have the highest confidence in this location—since it can be tied to an existing structure). Channel gradient at this location is approximately 0.8 percent. Due to the maintenance associated with Freshwater Park, the terrain was relatively open and we were able to survey about 350 feet, compared to the ACE's 450 feet. We assumed that the elevation of the terrace on the left side of the channel (looking downstream) remained constant and could serve as a reference datum. Based on a comparison of the two cross-sections, it appears that the channel has degraded at least two feet, and that the sediment has been moved downstream from this location (Figure 8).

Conclusions

The conclusions that can be drawn from this project are limited due to the approximations and assumptions necessary for remeasurement. If the original survey notes could be obtained from the ACE, these three cross-sections, and perhaps the other six that were established in 1975, could be surveyed with considerably higher confidence. As further watershed studies and/or watershed analysis is completed in the Freshwater Creek basin, this information may be able to be obtained and further work completed to document channel changes that have occurred over the past 23 years.

Based on the qualifications stated above, our limited conclusions are that these cross-sections suggest that only minor channel aggradation may have occurred in the lower gradient reaches of Freshwater Creek, perhaps on the order of six inches to one foot. Some antidotal evidence presented to CDF over the past year has further suggested that this level of channel filling may have occurred. As part of a long-term instream monitoring program in the Freshwater Creek watershed, both well documented longitudinal and cross-sections should be established and monitored over several decades to provide verifiable data regarding sediment movement and storage in the lower part of this basin.

References:

Dunne, T. and L.B. Leopold. 1978. Water in environmental planning. W.H. Freeman and Co., San Francisco. P. 605-606.

U.S. Army Corps of Engineers. 1975. Flood plain information, Freshwater Creek, Humboldt County, California. Final report prepared for Humboldt County by the U.S. Army Engineer District, San Francisco, CA. 14 p.

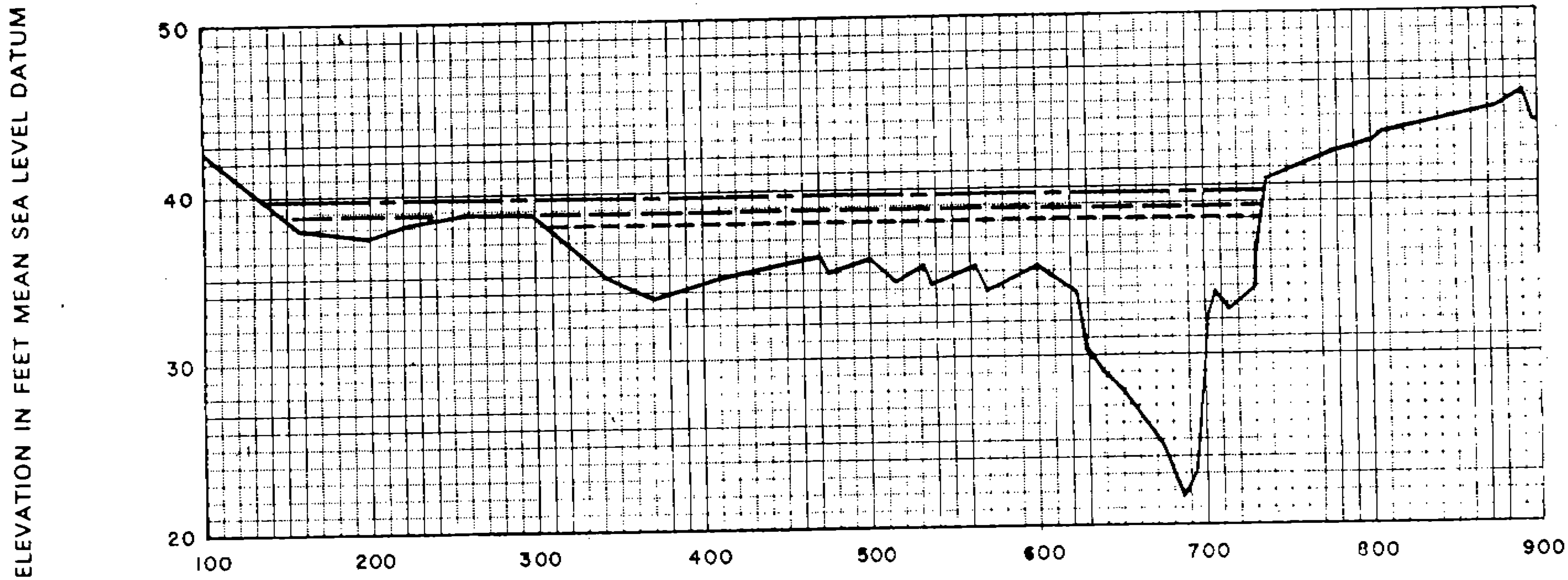
Peter H. Cafferata

Peter H. Cafferata
Forest Hydrologist
Sacramento Headquarters

Hugh Scanlon
Forester I
Humboldt-Del Norte Ranger Unit
Fortuna

cc: Mr. Dean Lucke, CDF Santa Rosa
Mr. Tom Osipowich, CDF Santa Rosa
Mr. Tom Spittler, CDMG Santa Rosa

Figure 1



CROSS SECTION NO. 3
RIVER MILE 2.74

LEGEND

—— STANDARD PROJECT FLOOD
— — — INTERMEDIATE REGIONAL FLOOD
- - - - - - 50 YEAR FLOOD
SECTIONS TAKEN LOOKING DOWNSTREAM

DEPARTMENT OF THE ARMY
SAN FRANCISCO DISTRICT, CORPS OF ENGINEERS
SAN FRANCISCO, CALIFORNIA
FLOOD PLAIN INFORMATION
HUMBOLDT COUNTY
CALIFORNIA
SELECTED CROSS SECTIONS
FRESHWATER CREEK
OCTOBER 1975

Figure

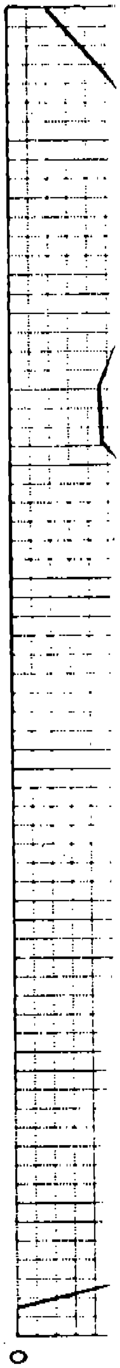


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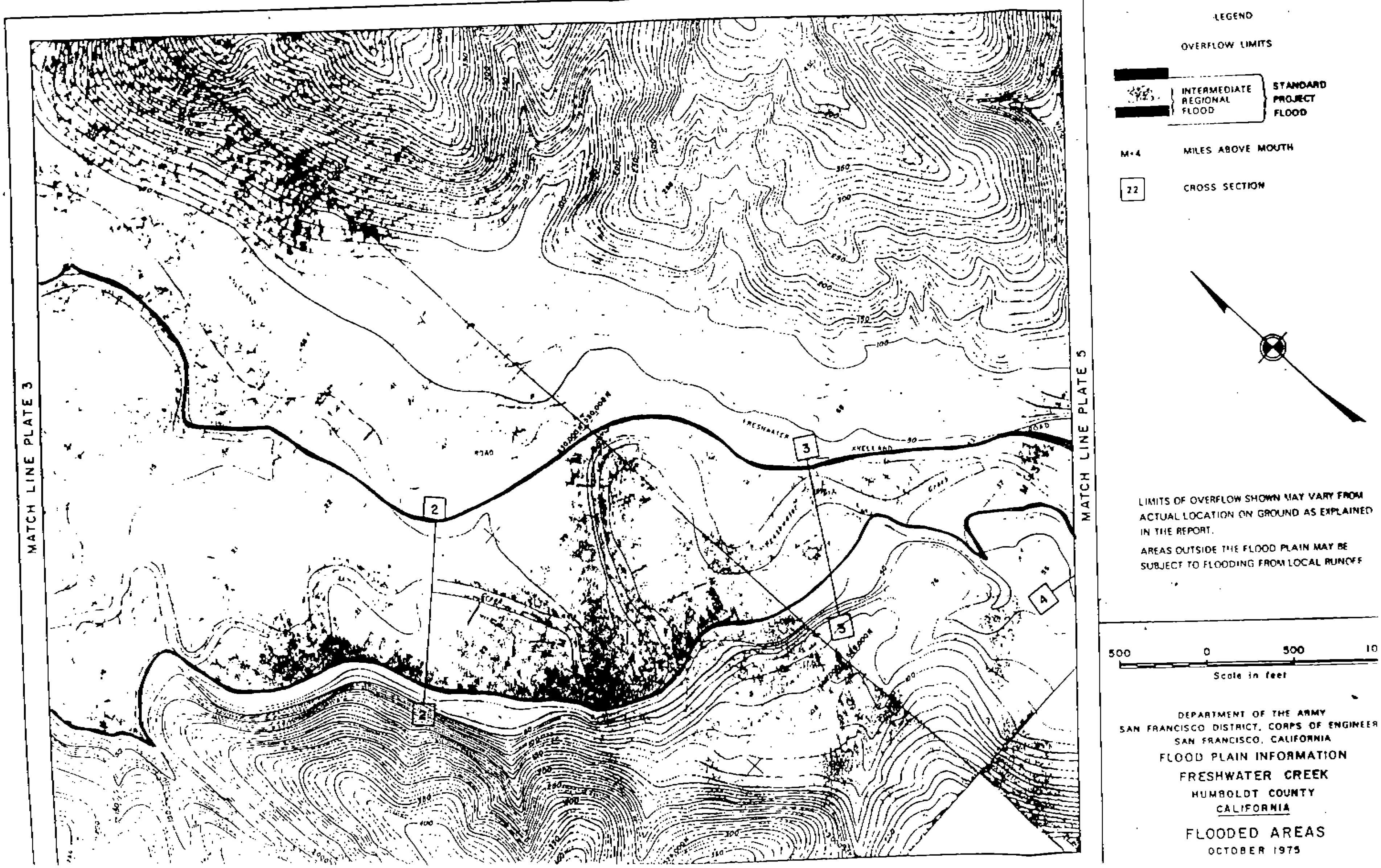
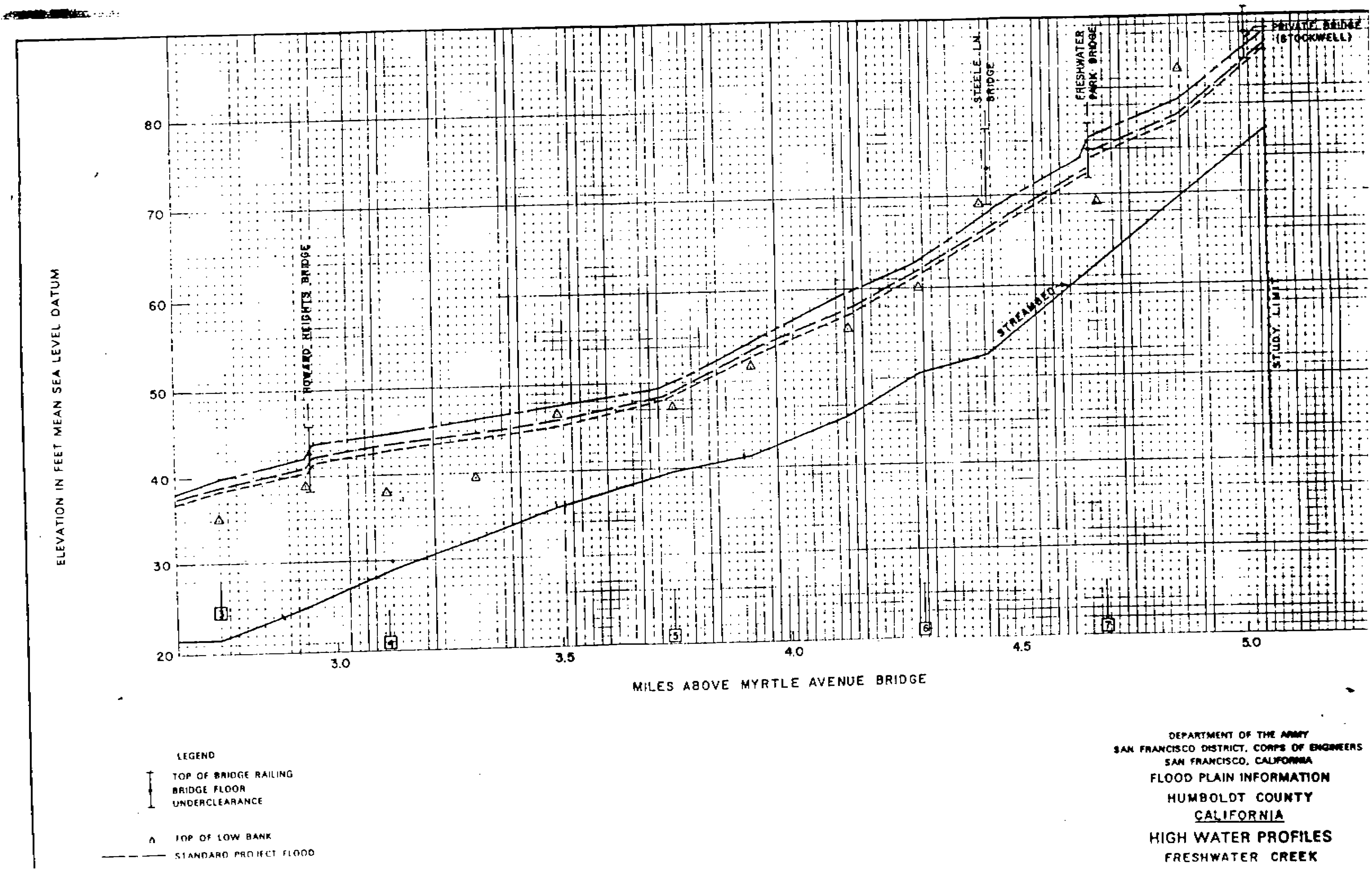
Figure 3



MATCH LINE PLATE 3
MATCH LINE PLATE 5
FRESHWATER
KNELLAND
ROAD
2
3
4
5
LEGEND
OVERFLOW LIMITS
INTERMEDIATE REGIONAL FLOOD
STANDARD PROJECT FLOOD
M+4 MILES ABOVE MOUTH
22 CROSS SECTION
LIMITS OF OVERFLOW SHOWN MAY VARY FROM ACTUAL LOCATION ON GROUND AS EXPLAINED IN THE REPORT.
AREAS OUTSIDE THE FLOOD PLAIN MAY BE SUBJECT TO FLOODING FROM LOCAL RUNOFF
500 0 500 10
Scale in feet
DEPARTMENT OF THE ARMY
SAN FRANCISCO DISTRICT, CORPS OF ENGINEER
SAN FRANCISCO, CALIFORNIA
FLOOD PLAIN INFORMATION
FRESHWATER CREEK
HUMBOLDT COUNTY
CALIFORNIA
FLOODED AREAS
OCTOBER 1975

Figure 4



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SAN FRANCISCO, CALIFORNIA

FLOOD PLAIN INFORMATION
HUMBOLDT COUNTY
CALIFORNIA

HIGH WATER PROFILES
FRESHWATER CREEK

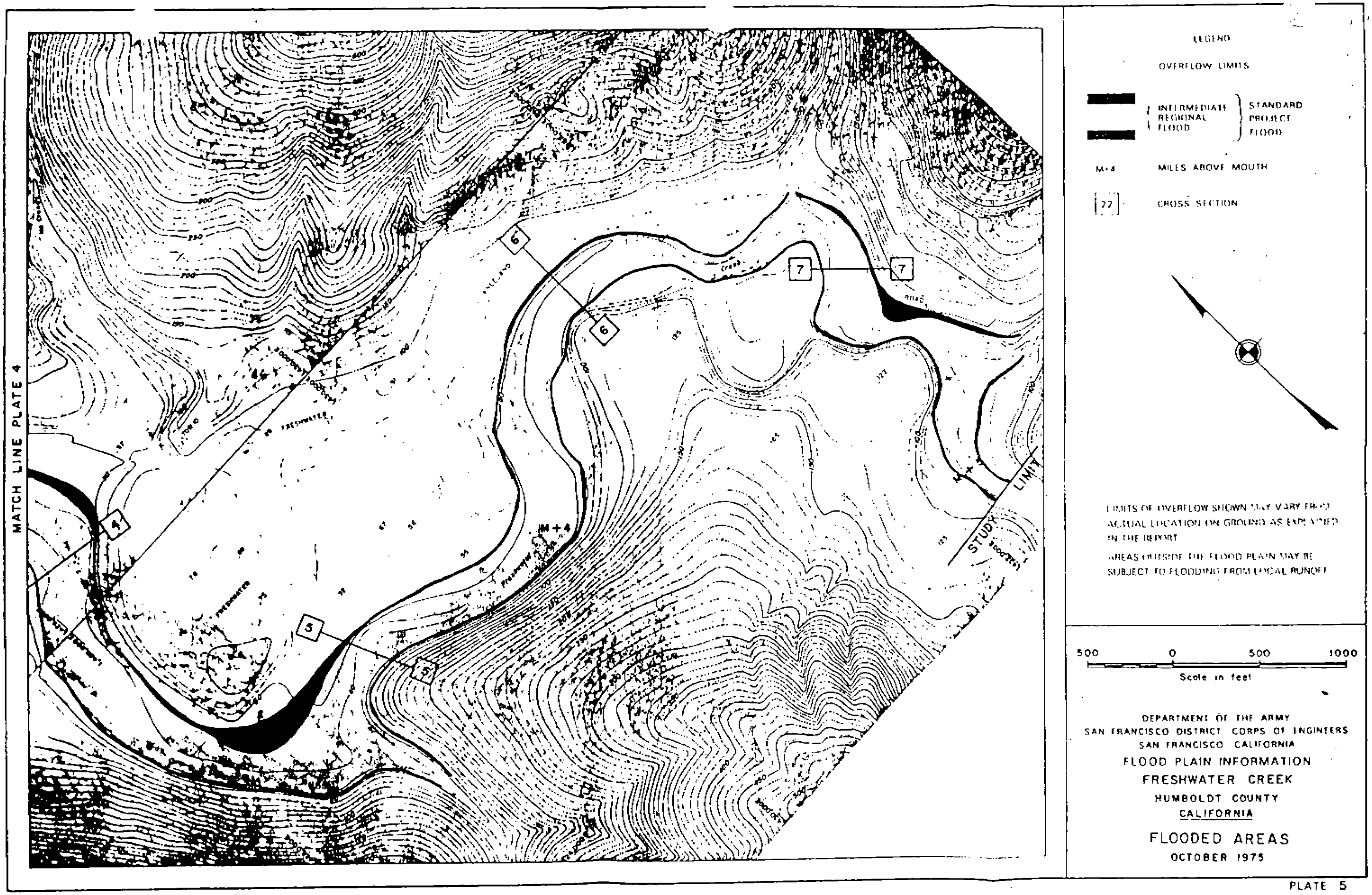

MATCH LINE PLATE 4
LEGEND
OVERFLOW LIMITS
INTERMEDIATE REGIONAL FLOOD
STANDARD PROJECT FLOOD
M+4 MILES ABOVE MOUTH
22 CROSS SECTION
LIMITS OF OVERFLOW SHOWN MAY VARY FROM ACTUAL LOCATION ON GROUND AS EXPLAINED IN THE REPORT
AREAS OUTSIDE THE FLOOD PLAIN MAY BE SUBJECT TO FLOODING FROM LOCAL RUNOFF
500 0 500 1000
Scale in feet
DEPARTMENT OF THE ARMY
SAN FRANCISCO DISTRICT CORPS OF ENGINEERS
SAN FRANCISCO CALIFORNIA
FLOOD PLAIN INFORMATION
FRESHWATER CREEK
HUMBOLDT COUNTY
CALIFORNIA
FLOODED AREAS
OCTOBER 1975
PLATE 5
STUDY LIMIT
M+4
4
5
6
7

Figure 6

Figure 7

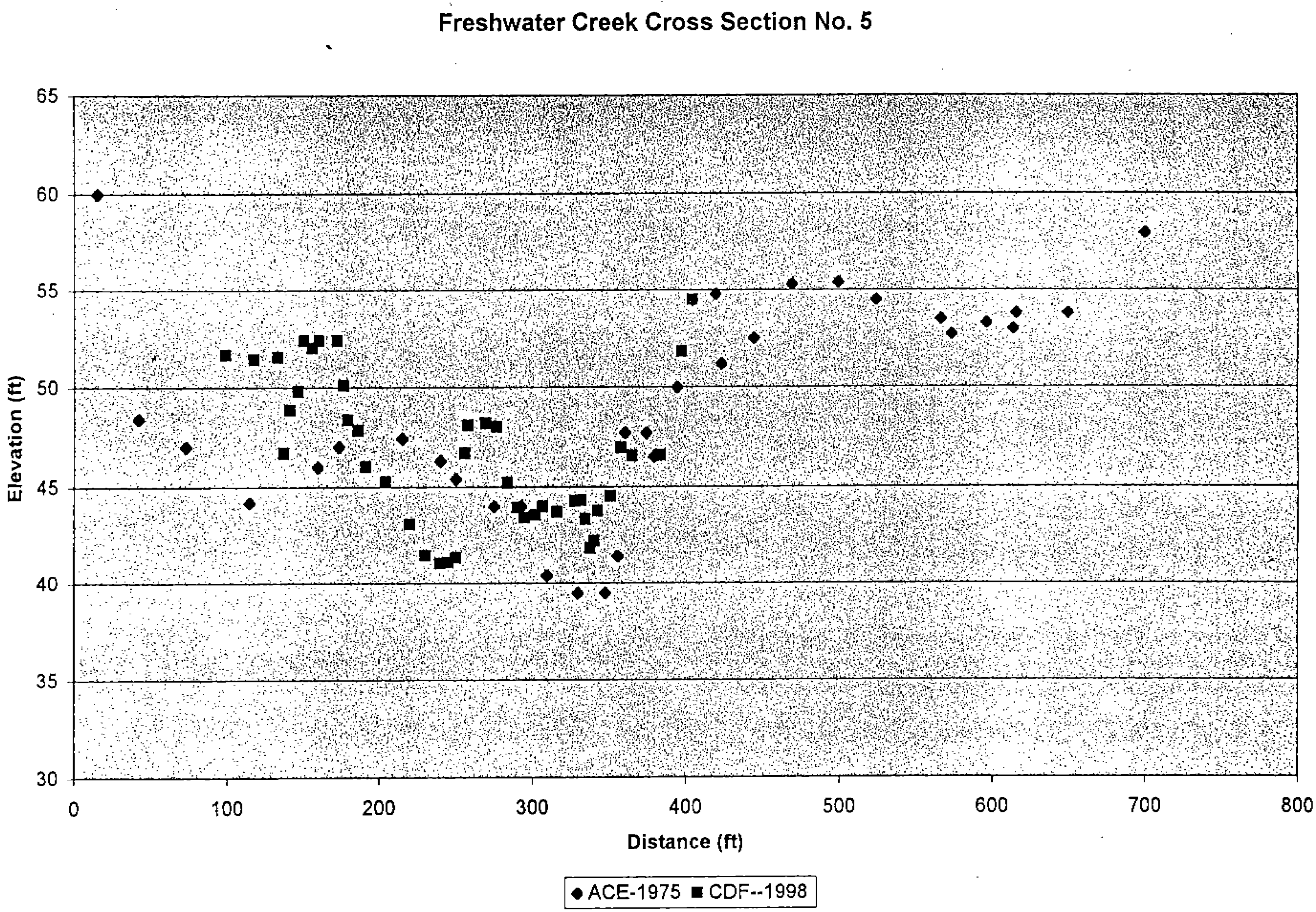
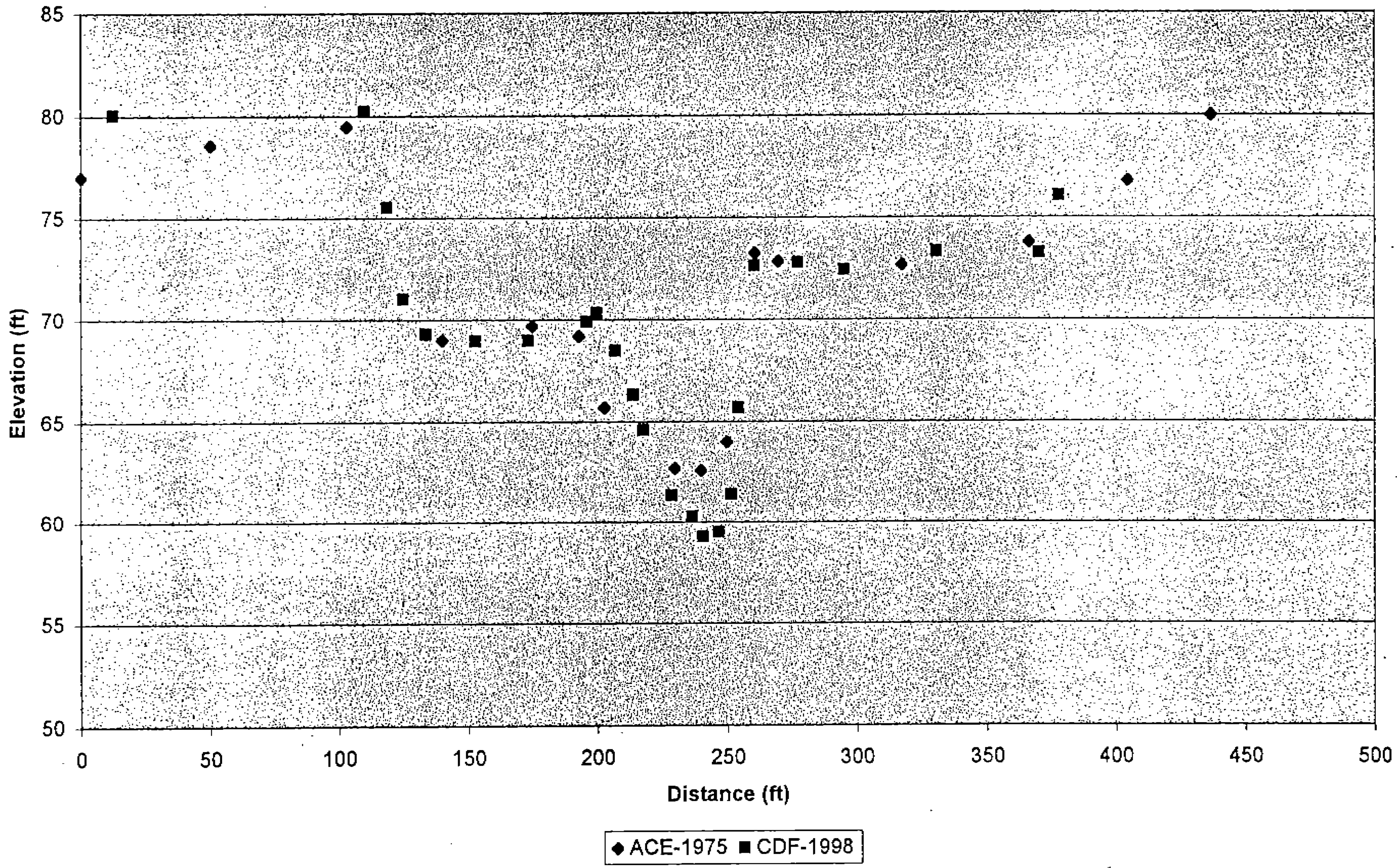

Freshwater Creek Cross Section No. 5
Elevation (ft)
65
60
55
50
45
40
35
30
0
100
200
300
400
500
600
700
800
Distance (ft)
ACE-1975
CDF--1998

Figure 8

Freshwater Creek--Cross Section No. 7



Attachment II-1

Dr. William Weaver's Response to Dr. Leslie Reid's Review Comments of the PWA Bear Creek Sediment Source Report

PACIFIC WATERSHED ASSOCIATES

Date: November 10, 1998

To: Dan Opalach, Ph.D.
Pacific Lumber Company

From: Dr. William Weaver, PWA
Danny Hagans, PWA

Subject: **Response to Dr. Leslie Reid's review comments of the PWA Bear Creek Sediment Source Investigation and Sediment Reduction Plan**

Introduction

In late October, 1997, the Regional Water Quality Control Board asked Pacific Lumber Company (PL) to prepare a technical and monitoring program report for the Bear Creek watershed. In late November, Pacific Watershed Associates (PWA) prepared a plan-of-action and schedule for work to be undertaken from December through March. It should emphasized that field work was accomplished using four to seven field personnel and included road erosion inventories, hillslope inventories, stream channel erosion inventories and landslide inventories. Field work was conducted during the winter period, and occurred concurrently with analysis of stereo aerial photography from 1947, 1966, 1974, 1994, 1997. Data analysis of field measurements and aerial photo interpretation occurred under a strict time line in order to prepare the final report by the deadline eventually imposed by the Water Quality Control Board.

The Bear Creek investigation and report (PWA, 1998b) was designed and undertaken to provide an increase in our knowledge of watershed sediment production and yield, specifically for the Bear Creek watershed. From our work in other watersheds and river basins of the north coast region, it is abundantly clear that the data and findings from Bear Creek are highly specific to that drainage, and not necessarily transferrable to other watersheds. As was later discovered, many of the findings do not even readily transfer to the adjacent Jordan Creek watershed where many natural conditions (geology, aspect, slopes, etc) are very similar to Bear Creek. Speculating on cause-and-effect, transferring data or findings, or developing conclusions about the operation of another watershed, based on Bear Creek data, is not warranted and will likely lead to erroneous conclusions.

In preparing the Bear Creek report, we developed a general statement of associations between landsliding and various natural conditions and management activities. For example, we listed the frequencies of recent (1997) landslides on slopes exhibiting various harvest ages. It is far too simple to assume that these associations are indicative of a cause-and-effect relationship, and they cannot correctly or accurately be employed in that manner. For example, each of these slope areas have been influenced by complex, multiple land use histories and multi-variate natural associations that have not been elucidated by this brief field study. Only with road-related sediment sources can the link between land management (road location, road design, road construction or road maintenance) and sediment yield be clearly established.

Geomorphic Studies • Erosion & Sedimentation Processes • Wildland Hydrology • Erosion Control
P.O. Box 4433 • Arcata, California 95518 • Ph 707-839-5130 • Fax 707-839-8168 • pwa@northcoast.com

As such, the Bear Creek study was undertaken within a strictly limited time frame, during the dead of an extremely wet winter period, and was designed to increase our understanding of how the Bear Creek watershed responded to recent storms. The project was undertaken as a management study, not as a research project designed for peer review and future publication. Additional tasks which might have been undertaken to refine the results, and more detailed analysis procedures which might have been employed to refine the basic conclusions would be the subject of another study. It is our view that many of the review comments of Dr. Reid should be viewed in this context.

We have conducted research on erosional processes in northern California over the last two decades, some of which has been formally reviewed by Dr. Reid. Had time been available, PWA would have solicited outside review of the Bear Creek report prior to public distribution. We are disappointed that Dr. Reid failed to acknowledge the intent and purpose of the study and the constraints under which the data was collected, interpreted and reported.

General comments

Dr. Reid (reviewer) has listed our four "stated goals" of the Bear Creek report. Some additional clarification on our objectives is probably warranted. As stated in the report, we identified and inventoried sources of erosion only if they resulted, or would result, in sediment delivery to a stream channel. Thus, for example, cut bank erosion and failures which did not deliver sediment to a stream were not included in the inventory and analysis. In this respect, the study was technically an inventory of sites of sediment delivery rather than an erosion inventory. Thus, virtually all sediment volumes listed in tables of the report (unless otherwise indicated) are sediment yield or delivery volumes.

<u>Review comment 1.</u> *The surface erosion component of the sediment source inventory appears to be significantly underestimated. (See also 22/4)*

We disagree. It should be noted that our estimate of surface erosion from roads is expressed as a fraction of the total delivery of sediment to the channel system. We did not estimate "surface and rill erosion rates," nor did we attempt to include surface erosion from skid trails as was stated by the reviewer. Thus, we did not include surface erosion from road reaches that are not somehow connected to streams (either through gullies or direct roadbed and ditch contribution). Because of the high relative landslide contribution in the watershed (82% of total yield), we estimated that surface erosion accounts for less than 2% of total delivery. This is a fair and perhaps liberal estimate that could be refined with additional work. It was felt that additional effort was not warranted.

More importantly, defining the extent to which road prisms, cutbanks and ditches have the physical ability to deliver sediment to stream channels (ie., their connectivity) should be the focus of the question. The analysis of roads through out the watershed indicated that only 14% of the road system currently has the potential to deliver runoff and sediment to streams.

Review comment 2. *The report should have compared the rate of landsliding in Bear Creek to old growth areas of Bull Creek rather than to 1947 photos of unmanaged Bear Creek. This does not provide a valid estimate. (See also 11/2)*

Based on a qualitative analysis of the topography and geology of remaining old growth areas in the Bull Creek watershed, it was not felt that comparing landslide rates of Bull Creek and Bear Creek would be productive. It did not appear that the areas were comparable. Inferring that the Bull Creek hillslope areas represent the response of an unmanaged Bear Creek would lead to incorrect conclusions. Rather, it was felt that reviewing pre-management photography of Bear Creek would provide a glimpse of the nature and magnitude of unmanaged large landslide frequencies for the basin, which could then be compared to large landslide frequencies in the same watershed under a managed state. As the reviewer suggests, this has limitations in that the storm magnitudes for the two periods are undoubtedly different. At the same time, it has the advantage of eliminating differences in watershed characteristics that would be encountered if a different area were selected for comparison. For example, approximately 90% of the Bull Creek watershed is underlain by Yager formation sedimentary rocks rather than older Coastal Belt Franciscan Complex bedrock geologies which dominate in Bear Creek. The differences between the two geologies are important.

Perhaps a more relevant and fruitful exercise would be to compare the pre-management (1947) landslide frequencies in both areas to ascertain the inherent susceptibility of each area to mass wasting (since they would have both been subject to the same storms, and neither would have the complicating effect of land management). If that were done, there would be a clearer basis for determining the relevance of potential differences in watershed response that might be observed in the post-management period.

Review comment 3a. *"The report details a plan to reduce road-related sediment inputs, but these were calculated to account for only 8% of the anthropogenic sediment input."*

Road-related sediment inputs are, by definition, anthropogenic. However, it is grossly in error to infer that all measured sediment inputs in the watershed are anthropogenic in nature. As the pre-management aerial photography reveals, landsliding was an important process in the watershed prior to management activities, and an undetermined number of post-management landslides can be logically inferred to have been unrelated to human activities. Road-related sediment inputs, therefore, represent 8% of total sediment inputs in the basin (anthropogenic plus natural).

Road-related sediment sources are relatively straight-forward to treat through control and prevention measures. Once a road has been inventoried, future sediment sources can be effectively and cost-effectively addressed at any time prior to the occurrence of the erosion. A relatively high percentage of the identified future sediment sources comprising this 8% of the total basin sediment delivery are amenable to prevention or control. Even at 8% of basin-wide sediment delivery, we believe this to be an important source of controllable sediment that can be cost-effectively addressed.

In contrast, landslides which are going to occur on hillslopes in the watershed, whether or not they are harvest-related, are (for the most part) not amenable to control or prevention. We cannot go back to unmanage the slopes or change the style of management that has already occurred. Unlike road-related erosion (over which we have great control), there is frequently nothing that can be cost-effectively done to diminish landslide rates or to prevent specific landslides which may be related to past harvesting.

Future harvesting-related landslides can best be addressed through the practices of avoidance and altering harvesting prescriptions at susceptible hillslope locations. Both techniques, avoidance and land use modification, first require recognition. If the point of initiation of future slope failures could be reliably identified, these sites could then be managed for reduced risk. Such management can take two basic forms: 1) generic identification of potentially susceptible sites, followed by exclusion, limitation and/or modification of operations in all the areas which match these sites, or 2) site specific identification and analysis of potentially susceptible sites at each proposed area of land use activity, and the prescription and implementation of unique limitations and/or restrictions on proposed activities at those locations.

The former, a type of hazard zoning, has the advantage of broadly encompassing most or all potential sites of future significant slope failure (e.g., all inner gorge slopes) but has the disadvantage of including a large number of sites which are not likely to fail and could otherwise be safely managed. The latter strategy, that of site specific landslide hazard identification, narrows the zone of exclusion to areas specifically identified as hazardous in the field, but runs the risk that potentially unstable sites may be overlooked, through error or through lack of field indication. Even in this case, because of the imprecise nature of the science, areas will also be protected that would not otherwise fail.

Both strategies assume that simple vegetation removal, either partial or complete, is having a significant adverse effect on slope stability and that deferring, modifying, or avoiding harvest will lead to a lower incidence of landsliding. The basic assumption may be true, but the strength of the relationship, and the ability to limit landslide frequency or size through vegetation manipulation in Bear Creek, is not known or well understood. Regardless, although such a research project might be fruitful, that data collection and analysis effort was well beyond the scope of this investigation.

<u>Review comment 3b</u>. *The implications of the observed landslide distribution for the identification of appropriate silvicultural practices were not explored. Further, it is stated that the mass wasting avoidance strategy will not be sufficient to "adequately reduce rates of landsliding during future storms," because:*

1. *Shallow debris slide sites are hard to recognize in the field*
2. *The 400' leave strip width does not include all landslides identified in the study*
3. *Inner gorges do not include slopes <65%*

In spite of the review comment, considerable discussion was included in the text of the report regarding observed landslide distribution in the Bear Creek watershed and its implications for both past and future management activities. As a result of our observations, we have recommended that new roads not be constructed on steep inner gorge slopes, and that certain roads that currently exist in these locations be considered for decommissioning. We have further recommended that steep inner gorge slopes be individually inspected by geologists for geomorphic indicators and hillslope conditions that are indicative or characteristic of potentially unstable or "sensitive" hillslope locations. We did not provide specific guidance in the report, but such conditions would include a host of field indicators including slope shape, slope steepness, slope position, and both hydrologic and vegetative indicators that are well known to competent field geologists trained and skilled in landslide and sensitive slope recognition.

It is unclear what Dr. Reid believes is an "adequate" reduction in landsliding. We believe that most susceptible sites can be identified by careful and thoughtful field inspection of inner gorge and stream side slopes. Our air photo analysis and field inventory showed that this is where most landslides developed. It is then only a matter of arming the field geologist with the ability and authority to modify silvicultural prescriptions, to delineate avoidance zones, to defer harvesting or to omit potentially unstable hillslopes from harvest plans where field conditions warrant such protection. We believe the Mass Wasting Avoidance Strategy explicitly provides such authority. Implementation of appropriate protective measures, ranging from harvest modification to complete avoidance, should prevent as many harvest-related mass failures as would any other available strategy.

Shallow debris slide sites, and features indicative of such sites, should be generally identifiable in the field. The goal is not to identify which sites will actually fail (a nearly impossible task) but to isolate those sites which contain well known geologic, geomorphic, vegetative and hydrologic indicators of pending or potential instability. Actual signs of instability need not be present for a geologist to identify a site as potentially unstable. In fact, using such field criteria, the inspecting geologist is likely to identify many more potential sites than would actually fail in a storm. This conservative assessment should actually result in the protection of more slopes than would be expected to fail. As with any assessment, some sites may not be accurately predicted, but our experience suggests that most will.

The proposed Mass Wasting Avoidance Strategy (MWAS) does not differ significantly from the approach or standard currently employed by the USFS in Northern California. Prior to, or as a part of timber sale layout, qualified geologists stratify the watershed or applicable lands into several different geomorphic terrains. They then field sample to verify the air photo analysis and stability predictions. It is at this step, the field inspection to determine the risk of failure, that geologists and foresters discuss and negotiate, in the I.D. team meeting, a final solution to how the particular hillslopes will be managed, mitigated or avoided.

The MWAS mirrors recommendations made by the USFS, Klamath National Forest, in its "Salmon Sub-Basin Sediment Analysis" (De La Fuente and Haessig, 1994). In Chapter 9, Section B, "Prevention of Future Management-Associated Landslides and Erosion," the USFS recommends standards for earth science investigation:

> 1. "Develop and implement standards and guidelines for managing the sensitive geomorphic terranes and soil types identified by this study. Certain terranes and soils account for a disproportionately large portion of the total sediment delivery to the river system."
>
> 3. "Conduct appropriate geologic, soils and hydrologic investigations prior to conducting any significant soil or vegetation disturbing activities."

The 400 foot default no-cut strip established in the MWAS does not include all sites which have failed in the past, but it includes the great majority. For example, 41 of the 44 recent landslides in Bear Creek (those which occurred in 1997), would have been completely contained within the 400 foot maximum inner gorge slope length used for field analysis in the Avoidance Strategy. These 41 slides accounted for over 60% of the documented landslide yield from 1997 slope failures. Of the remaining three landslides, one was so large, and deep, that its failure may have been completely unrelated to recent harvesting. It alone accounted for 33% of the total 1997 landslide delivery to Bear Creek.

Finally, by our definition, and by that of the Mass Wasting Avoidance Strategy, inner gorge slopes include those which are at least 65% in slope gradient. These account for 70% of all landslide sites from all photo periods, and over 75% of the most recent (1997) landslides. We have recommended that, among other areas, inner gorge slopes be identified and inspected by field geologists. We would agree with Dr. Reid that lower gradient slopes in the stream side areas may also warrant inspection. In the database, 12% of all landslides occurred on these stream side slopes, and 15% of the 1997 slides occurred here. Based on this re-assessment, we would recommend that stream side slopes in the 50-65% category also be investigated in the field to identify sites of potential instability.

<u>Review comment 4.</u> *"Methods to reduce sediment delivery were discussed primarily with respect to road surface erosion, and consist of diverting road-surface runoff from the surface before it reached sites from which it could drain to a stream." This will aggravate slope stability problems. Rolling dips and outsloping would be difficult to sustain during periods of hauling on a wet roadbed, and maintenance would result in excessive sediment production (see also 31/6).*

We strongly disagree the Dr. Reid's assessment and believe she has missed the point of the road assessment and erosion prevention plan. First, a variety of methods were discussed to reduce sediment delivery, with road surface erosion being the least important (volumetrically) and least discussed. For road-related erosion we provided recommendations for excavation of unstable and potentially unstable sidecast fillslope materials, for the excavation of stream crossing fills on roads recommended for decommissioning, for the upgrading of stream crossing culverts where drainage facilities are currently under-designed and at risk of failure, for the elimination of stream diversion potential at all stream crossings on roads planned for retention, and for a variety of other erosion prevention tasks aimed at reducing the risk and volume of future road-related sediment delivery. None of these treatments are aimed at "road-surface erosion."

Our only recommendation for the control of road surface erosion (which we believe to be a minor component of overall watershed sediment delivery), is to disconnect the road's ditch system from the adjacent stream network. This is best accomplished by the installation of rolling dips and/or ditch relief culverts that break up road surface runoff and disperse it uniformly across the landscape, rather than allowing it to collect, concentrate and be discharged into stream crossing culverts. This practice is considered state-of-the-art road surface drainage techniques and is supported by both local and regional research (e.g., Wemple, 1994). If roads are outsloped, runoff and fine sediment is uniformly discharged across the hillslope. Where individual sites of road surface runoff are drained and discharged through ditch relief culverts or rolling dips, drainage sites and spacing can easily be chosen to minimize or eliminate the potential effect of the runoff.

Finally, used in appropriate locations, outsloping and rolling dips are not difficult to maintain on active haul roads. They are considered state-of-the-art, low cost and low maintenance road surface drainage practices throughout the northcoast region. We have prescribed only 490 feet of road outsloping in the entire watershed (0.2% of the 39 mile road network). Instead, we have concentrated on the use of ditch relief culverts and rolling dips (n=63) to improve and disperse road surface runoff. Any road maintenance practice can result in unnecessary sediment production and delivery to stream channel if they are incorrectly performed. The benefit of disconnecting the road surface and ditch from the stream system (using rolling dips) is that runoff and fine sediment that is generated from road maintenance activities and wet weather hauling will not be delivered to the channel system.

Assessment of major findings

Review comment #1. *"The report does an admirable job of identifying the dominant influence of silvicultural practices on slope stability in [the] Bear Creek watershed."*

Unfortunately, we were unable to provide the clear picture of causative links between specific silvicultural practices and landsliding described by Dr. Reid. The entire watershed had been recently logged by the mid-1960s so that the comparison of effect of first cycle logged and unlogged hillslopes could not be identified in Bear Creek. With the second major storm period in the late 1990s, second cycle logging had occurred on just over one third of the basin. Although we agree that elevated landslide frequencies on these recently harvested slopes in Bear Creek suggest a relationship between harvesting and slope instability, insufficient time was available to research and determine the influence of specific silvicultural practices.

In fact, as was made clear in the adjacent Jordan Creek watershed, it is unclear what role harvesting actually played in the observed landslide frequencies. In 1997, 41 landslides occurred on hillslopes in the Jordan Creek watershed. Slopes in Jordan Creek were nearly equally divided between those that were logged less than 15 years ago and those that were harvested over 30 years ago. In spite of this, and in direct contrast to landsliding in Bear Creek, 85% of the landslides occurred on the older harvested areas. The role of silviculture in contributing to hillslope landslide frequencies during storm events is not as clear as suggested by Dr. Reid.

Review comment #2: *"The report... documents the relative unimportance of road-related sediment sources in the area."*

Road-related sediment sources account for at least 10% of basin-wide sediment delivery from all sources, both natural and anthropogenic. The Bear Creek report indicates that this value is to be considered a minimum, and that sediment yield from eroded stream crossings may be underestimated by 10% to 25%. Dr. Reid further suggests that road surface erosion could be significantly higher than reported, if her rough calculations are correct. Finally, we have revisited some of the air photo landslide data in the original report and have shifted some landslide volumes from the hillslope category into the road-related category. Regardless of its actual contribution, we consider road-related erosion to be an important sediment source, if for no other reason than the fact that most of the road-related sediment sources can be easily and cost-effectively prevented or controlled by the application of road upgrading or decommissioning measures.

Review comment #3: *"Several other conclusions..[regarding pre- and post-logging landslide frequencies] are not supported by the data presented."*

Large storms are generally acknowledged to be important triggering events for increased landslide activity in wildland watersheds. Our data and observations confirm this finding. Many slides occur during the largest storms, and these events are typically followed by years of relative landslide quiescence. The only available tool or evidence which can be used to define the occurrence of past landsliding and landslide frequency is found in periodic aerial photography that is available for these watersheds. If photos were available for every year, then the role of individual storms would be easier to evaluate.

Photos in Bear Creek are available for the years 1947, 1966, 1974, 1994, and 1997, yet these photo years do not closely "bracket" the suspected triggering storms of 1955, 1964, 1972, 1975, 1995, 1996 and 1997. Major triggering events appear to have occurred in 1964 and 1997. We used data from these photos to calculate landslide frequencies for various time periods, including the prelogging period before 1947 and the most recent period beginning with the 1966 photos and ending with the storm of 1997.

Dr. Reid asserts that landslide frequencies calculated in the report are not correctly evaluated, and suggests that frequency should be calculated on the shortest time period for which data is available. She offers her own stab at a frequency calculation. However, her calculations suffer from the same inherent limitations: that is, most landslides are storm-triggered and do not occur over an interval. By picking two time intervals of vastly differing lengths, (50 years versus 3 years for pre-1947 and 1994-1997, respectively) she has artificially increased the frequency of sliding in the most recent period. It is probably best to calculate landslide frequencies over similar time periods and over time periods containing similar storm-triggering events. This is not straight forward, and is naturally limited by the availability of the photographic record and our limited knowledge of the hydrologic characteristics of past storms in the basin.

To make a more direct comparison, Dr. Reid suggests comparing landslide frequencies on lands with different harvest histories within the watershed during the same time period. This minimizes the effect of unknown differences in storm intensity and record length. In the Bear Creek report, we have performed this calculation and discovered a relatively high frequency of 1997 landslides on recently harvested hillslopes. However, data from the nearby Jordan Creek watershed, where unit landslide frequencies were much higher on first cycle logged lands than on recently logged lands, directly contradicts this interpretation. It is clear from these results that the extension of preliminary, "visual" relationships or associations drawn from this management study is not warranted, regardless of how comparable or how close those basins are to Bear Creek or Jordan Additional, more definitive research is needed to fully test the relationship between harvesting and landsliding before the magnitude of the effect can be evaluated and before the results can be responsibly applied to other areas.

Review comment #4: It is not valid to conclude that implementation of the Forest Practice Rules (FPR) have led to measurable and significant reductions in long term sediment yield, largely because PWA computed yields that included a period after the implementation of the FPR that had little harvesting (and hence would have received little benefit from the new procedures).

Our complete statement in the Bear Creek report was actually that "the Forest Practice Rules, as well as recent changes in road location, road construction and harvesting techniques employed by the landowner (e.g.., increased emphasis on ridge-top road location, and the utilization of cable yarding systems) are having a measurable and significant effect in reducing long term sediment yields." It is our observation that observable changes in practices, implemented since the inception of the FPRs, prevented erosion and sediment delivery during the storm events of the 1990s, compared to what would have occurred under the less protective measures that were employed in the pre-FPR period. At this point it is not possible to quantify the magnitude of this reduction.

Dr. Reid's comment regarding the applicable benefit period (regarding the evaluation of the FPR) has merit, and with unlimited time we could have added several more time periods to the air photo analysis depicted in Table 8. However, we did clearly separate out the 1994 to 1997 sediment delivery volumes in Tables 2, 3, 5, 6 and 7. The rate for the three year period (21,230 t/mi^2/yr) was stated in the text of the report (p. 25) and perhaps missed by Dr. Reid.

Review comment #5: A more direct comparison of landsliding rates during active first-cycle and second-cycle logging would be a comparison of average landslide frequencies per unit area of land logged per landslide generating storm (see also 26/2).

For the reason stated by Dr. Reid, that additional landslide producing storms occurred during the periods in question, her rough calculations (26/2) and conclusions of the differences between landsliding rates from first- and second-cycle logging are not particularly meaningful. In addition, Dr. Reid was critical of the Bear Creek report excluding the single large "outlier" landslide (identified from 1997 aerial photos) in the analysis of large and very large landslides. However, she goes on to calculate first-cycle unit sediment delivery both with and without this large slide and concludes that "volume rates of sediment input [for second cycle logging] are only slightly lower." Even using her calculations, unit rates of sediment input from the watershed after first-cycle logging are 140% of second-cycle logging. This appears significant.

In addition, subsequent analysis of storm magnitudes for events in the 1990s, both in the lower Eel River basin as well as in the Humboldt Bay area, suggest that recent storms were of significantly greater magnitude than originally believed. Preliminary USGS gaging data in the nearby Bull Creek watershed has ranked the 1996 event as the storm-of-record, surpassing even the legendary floods of 1955 and 1964 (Cafferata, written communication, 12/10/97). The magnitude of the consecutive two-month precipitation totals for three individual storm periods at Scotia (one each in 1994-95, 1995-96 and 1996-97) were in the top ten for the 72 year period of record at that station, and all exceeded that of 1964 (CDWR, 1998). Of 46 storm periods recorded at Kneeland, east of Eureka, over the last 47 years, three of the largest events occurred between 1994 and 1997 (Conroy, written communication). All these lines of evidence point to the occurrence of geomorphically significant storms in the 1990s.

However, these concerns aside, Dr. Reid's suggested approach has merit if all the factors are considered in the analysis. Time constraints did not allow for such an analysis initially, but it may provide some interesting and useful insights in the future. The procedure will still be plagued with several inherent difficulties or limitations. First, such a comparison would require the assumption that storm magnitude and duration for the two time periods were comparable (or at least were known) in terms of their ability to generate hillslope landslides. It is unlikely that such data exists, so we would need to establish storm characteristics from rainfall and runoff data elsewhere. This throws some additional uncertainty into the results. In addition, because of a lack of bracketing photography, it will not be possible to separate the effects of individual storms (for example, 1955 from 1964) on landslide generation. Second, this analysis would not account for differences in silvicultural and yarding practices during first and second-cycle harvesting periods. It is unclear if this potentially important effect could be evaluated.

Review comment #6: Because soils and bedrock similar to those in Bear Creek are also found elsewhere in the Mattole, the lower Eel and in parts of Freshwater Creek, "information and conclusions drawn from the Bear Creek watershed are thus potentially relevant through a large area."

Dr. Reid is careful with her words and suggests, because of similarities in geology and/or soils, the findings in Bear Creek are "potentially relevant" to a larger area. In the regulatory and public arena through which this report and its findings (and her review comments) can be expected to travel, and be used, this is a dangerous statement. It presents a misleading picture to both untrained regulatory agency personnel, industry personnel and land managers, and to members of the general public who wish to understand how this study may have relevance to other situations and other settings. Her statement provides no guidance as to what information and which conclusions she believes are "relevant" and, more importantly, how this relevancy can be translated by extension to other areas. This is not a simple problem, and one that is fraught with pitfalls and potentially misleading and incorrect application. Through the use of such unsupported statements and unqualified "guidance," bits and pieces of data, findings and conclusions are dragged across the landscape and employed in situations which may have little in common with Bear Creek.

As authors of the Bear Creek study and report, we strongly discourage the blind or untrained extension of results or conclusions from this single field study of one small watershed to other areas of the landscape. It is neither warranted nor justified to consider Bear Creek to be an analog to other areas, and to extend data or conclusions from this site to other areas. For example, in recently completing a similar sediment source analysis for the adjacent Jordan Creek watershed, which is remarkably similar in most ways to Bear Creek (certainly much more similar than the Mattole or Freshwater Creek) we discovered harvest and landslide associations that directly and dramatically contradicted those encountered in Bear Creek. It is clear that additional work and analysis is needed and that extension of data, results or conclusions is not yet warranted.

The Bear Creek sediment source inventory was the first in a series of watershed sediment source assessments that are being conducted on Pacific Lumber Company lands. It represents one of the most detailed analyses of sediment production and delivery, and associated geomorphic and land use associations, to be performed on a north coast watershed in over a decade. The report provides a preliminary look at the main sediment sources in one small drainage basin. Data and results from Bear Creek, as well as the results of other studies recently completed or now underway for nearby watersheds, will need to be carefully evaluated before it will be reasonable and warranted to apply specific findings or general conclusions to other watersheds.

Specific Comments

Dr. Reid's has provided review comments and suggestions under the title "Specific Comments." Some comments provide useful suggestions on how additional analysis may reveal trends and associations in the data. We have already discussed some of these topics. A few of the remaining comments are discussed below.

3/1 Virtually the entire Bear Creek watershed (except the extreme lower basin) is underlain by undifferentiated Coastal Belt Franciscan rocks. According to present geologic mapping, all slides occurred on the same rock type so a further evaluation of slide distribution is not possible.

5/5 The presence or absence of seismic activity preceding a storm is likely to have a direct but unquantified influence on the frequency of landslides which occur. If harvested areas and logging road fillslopes are located on more susceptible sites on the landscape, then one might expect a disproportionate response from these managed areas, compared to unmanaged or older managed sites following a seismic event. The occurrence of episodic seismicity and uplift, and its role in initiating slope instability or triggering landslides, is also important when data and conclusions on landslide frequencies and sediment delivery are extended to other areas which have a lower seismic component. Clearly, the preliminary association of landslide frequency and land use that we identified in the Bear Creek report are already finding their way into discussions of other watersheds. For this reason, it would not be appropriate to omit the general discussion relating the potential relationship between seismic shaking and landslide occurrence.

5/7 Unfortunately, as far as we are aware, no photo set exists covering the watershed between the 1955 and 1964 storm events. In selecting the photos for this analysis, we attempted to identify photo sets that would bracket major storms, to the extent they were available, as suggested in Reid and Dunne (1996).

6/1 Good suggestion. Evaluating land use history upslope from landslides, as a possible contributing factor to slope failure, would add another factor into the possible equation which explains why some slopes failed and others did not fail. It may also be useful to classify the nature of the harvest (clear cut, selection, etc.), as well as the style and intensity of harvesting that occurred on-site and upslope during the first-cycle logging. As can be seen, the role of each of these many complicating land use factors can become almost indecipherable. In our initial analysis, we simply differentiated between recently harvested (<15 years old), older harvested (15-30 years old) and advanced second growth areas (>30 years old) at the site of landslide initiation. Based on work in the nearby Jordan Creek watershed, it appears that this differentiation is probably not sufficient to provide a clear delineation of the cause and effect relationship, or even the true magnitude of the association between "harvesting" and landslide occurrence. This would require a much larger effort, and still may not meet with unambiguous results even for Bear Creek. Certainly, these preliminary results and conclusions should not be extended outside this particular watershed. There is certainly room, and a need, for additional and more detailed academic research on these topics. There was neither the time nor the mandate to fully develop this management project into such a research effort.

8/3 Cutting history data is available for each of the photo years that were analyzed for landsliding. We chose to lump harvest ages into the three age categories described above for our analysis. These equal age classes capture the basic periods of susceptibility due to root strength loss and could be roughly delineated on the available aerial photos.

10/3 The 1947 photos capture the watershed before significant land use. The 1954 photos, which cover most of the basin, show additional land use, a few small road-related landslides, and just one large landslide, associated with both harvesting and road construction on inner gorge slopes. Almost all the landslides that showed up on the 1966 photos did not appear on the 1954 photos. They were new and are assumed to have developed either in response to the 1955 or the 1964 storm events. There was no reason to perform an analysis of the 1954 photos. We have no photos between these storm events which would allow us to separate the effects of each storm independently.

11/2 As per our earlier discussion, a comparison of landslide rates between managed Bear Creek and unmanaged Bull Creek would not have been a valid comparison. The two areas contain different geologies and different topographic characteristics. We looked at old pre-management photography of both areas, and there is a clear and distinct difference in their susceptibility to landsliding. Rather, it might have been useful to compare signs of pre-management landslide activity in both watersheds to establish a relationship between the two drainages, and then to compare post-management landsliding. This might be a fruitful research project.

As with any air photo exercise, ages assigned to landslides are approximate. Data presented in Table 2 identifies inner gorges, steep streamside slopes and headwater swales, whether managed or unmanaged, as the most susceptible sites for landsliding in Bear Creek.

11/4 This is a valid point, and further field and air photo research is probably warranted to identify the relationship between the visual appearance of landslides in old growth settings, and their actual age as measured using increment boring in the field. This could be done in Bull Creek. Lacking quantitative data, we used professional judgement based on many hundreds of hours of experience analyzing aerial photography and landslides in both cutover and old growth settings in Redwood National and State Parks.

12/2 We did not feel there was a problem identifying landslides which occurred in the 1974 to 1994 time period. When needed, we have gone back to 1981 aerial photography to answer any questions we may have had about individual slides.

13/3 Our analysis did not include a quantitative delineation, and area computation, of inner gorge slopes in the Bear Creek watershed. Instead, we classified the terrain at the point of initiation of each landslide, according to hillslope gradient, geomorphic form and proximity to a major stream channel. We identified inner gorge and steep streamside slopes along the main stem of Bear Creek as well as along many of its major tributaries. We agree, it might be a useful exercise to formally delineate the location and extent of inner gorge slopes throughout the watershed, since these are the most sensitive slopes in the basin. Areal analysis of inner gorge and streamside slopes, and areal analysis of watershed harvest status through time, were tasks that we discussed but were precluded from completing because of time limitations. A similar, but multi variate, analysis was performed using GIS to identify susceptible landslide sites in Bear Creek and elsewhere. This analysis serves as one of the triggering mechanisms which is used to call a geologist to evaluate proposed timber harvesting plans in the basin.

In reference to Kelsey's work, we do not believe that it is useful to define inner gorge slopes in Bear Creek based on information derived from the Redwood Creek watershed. We have extensive knowledge of both basins, based on 20 years work in Redwood Creek and our detailed field assessment in Bear Creek. Conditions and classifications are not readily transferrable. We do not disagree with the idea that lower gradient slopes may also be susceptible to slope failure, and indeed, our inclusion of "streamside slopes" (which are geomorphically equivalent to inner gorge slopes, just less steep) encompasses the slopes which Dr. Kelsey defined as "inner gorge."

14/1-2 Statistical measures were not used to test for differences in mean landslide
17/2 dimensions from one time period to another. There was no attempt on our part to forward or test hypotheses related to landslide dimensions, so we did not invest the additional time to develop these measures. Instead, we reported the simple averages. We recognize that with small populations, a few large landslides can have a disproportionate effect on average dimensions and volume calculations. With additional time, we could have performed additional analysis and developed the descriptive statistics for the data set. Management reports designed for regulatory agencies and for land owner use are not typically written with descriptive statistics. The data could be further analyzed if these audiences desire the information.

15/1 Recent research and landslide inventories following the 1996 storms in coastal Oregon suggest just the opposite: that landslides are much more difficult to identify from the air and from aerial photography in areas with a vegetative canopy. However, we agree that it requires great care and experience to differentiate small landslides from bare ground in tractor yarded areas.

15/3 We agree, it is important to try to separate the effects of first-cycle logging from second-cycle logging. This is much more difficult than it may seem at first glance. First, all landslides that occur in areas that have not yet been reharvested are either the result of first-cycle effects, or they are natural. We do not believe it is valid to assume that all landslides which occur on harvested slopes are the result of harvesting. Determining what are reasonable background rates of landsliding then becomes important. And, as is discussed above, developing estimates of background landslide rates is not as simple as comparing Bear Creek with Bull Creek. Second, landslides which occur in second-cycle harvest areas cannot automatically be attributed to the most recent harvesting. Some are due to first-cycle effects, other to second-cycle effects and still others are natural failures which may not have been caused by harvesting effects alone. Differentiating these causes is not straightforward and was beyond the scope of the Bear Creek project.

17/2 We agree, some landslides observed on the 1966 aerial photos were indeed triggered by the 1955 storm. However, after analyzing the aerial photography, we remain convinced that most were triggered by the 1964 storm event. This observational conclusion is based on the amount of revegetation, gullying and post-failure "recovery" of the slide surfaces.

17/3 The point of Table 4 is to show that most of the sediment delivered to the stream system in Bear Creek has been derived from a relatively small number of large and very large landslides that occurred during episodic storm events. The time intervals were selected because these are the time periods when the largest landslides show up; right after major storm events. Intervening periods don't have large landslides. Regardless of the time period expressed by the pre-management (old growth) landslides (we estimated they were less than 50 years old), the 1947 photo data clearly shows that large and very large landslides occurred and were relatively common in the unmanaged watershed. The data further shows that these large slides also occurred in both of the selected management periods. The numbers and the volumes of these large landslides may be more a function of storm differences than management conditions. In fact, there are those that would argue that many of the largest mass movement features are natural and largely unaffected by vegetation manipulation. It is our opinion that management has played a role, but the magnitude of that influence is debatable and ripe for additional research.

17/5 Research in Redwood Creek, South Fork Trinity River and elsewhere in the north coast suggests that non road-related landsliding typically provides more sediment to stream channels than road-related failures. This is not a new or unique finding to Bear Creek, to this geologic type or to the north coast. Aerial reconnaissance inventories of landslides triggered by the 1996 storms in the Oregon and Washington Cascades and the Oregon Coast range suggests similar conclusions. There, we identified only 36% of 651 landslides as being road-related (PWA, 1996). In contrast, aerial surveys in Idaho revealed that 65% and 72% of the observed landslides in the North Fork Clearwater and Lochsa River basins, respectively, were associated with roads (PWA, 1998). The relative importance of road-related landslides varies from area to area. We would agree that the dominant anthropogenic influence in Bear Creek is probably related to silviculture.

18/4 Our initial conclusions about the relationship between silviculture and 1997 landslide frequencies was described in the Bear Creek report as "suggestive" but not definitive. Based on our subsequent inventory and analysis of landsliding in the nearby Jordan Creek watershed, it appears that the association and apparent relationship may not be as solid as it appears on the surface. There, landslides were much more common on older harvested hillslopes than they were on the more recently cut areas, even though both types of slopes were equally distributed in the watershed. It is clear that additional research is required before intelligent conclusions can be developed for Bear Creek and Jordan Creek. Similarly, because of the widely differing results between Bear and Jordan, it would be improper and inaccurate, in our opinion, to employ the inferred relationship to draw conclusions about watersheds and processes outside the Bear Creek watershed.

19/3 Most landslides do have depths in the range from 2 to 5 feet, and this is within the rooting depth of most conifer species. However, the large and very large landslides that contribute the great bulk of total sediment to the channel system of Bear Creek often display deeper failure depths. Decreased root strength following harvest may not be as viable an explanation for the occurrence of these features.

21/2 We did not perform a vegetation analysis of the main stem of Bear Creek from the 1947 aerial photography. Because of shading and scale, it might be difficult to determine species composition and the age of riparian vegetation. Additional investigation might be a worthy undertaking.

21/3 The base of the channel fill deposits was visible at many of the measured channel cross sections. Channel cross section measurement points were often specifically selected at locations where indicators of original stream bed could be identified. Still, stored sediment measurements should be considered to be a minimum value.

22/4 We have already discussed some or our assumptions and thoughts on the volumetric importance of road surface erosion in the overall sediment budget of the Bear Creek watershed. Without belaboring the facts, it is important to reiterate that there are only 5.5 miles of road in the watershed today that deliver road surface runoff (and sediment) to stream channels (either through ditch relief culverts, gullies or directly into stream crossing culverts). If we assume that surface erosion delivered 2% of the total basin sediment yield (or 37,300 yds^3), this represents the removal of over 1.5 feet of the road surface (assuming a 20-foot wide road). It is our opinion that, in this small watershed, the large contributions of mass movement dwarf the volume delivered by road-related surface erosion. Total yield from this source is probably substantially less than 2%.

25/3 Additional analysis of the magnitude of recent storms suggests that the storm events of 1995, 1996 and 1997 were larger than originally described in the Bear Creek report (see Assessment of Major Findings, review comment #5, above). We agree with Dr. Reid that storms are the triggering mechanism for landslide initiation in Bear Creek. It is therefore important to understand the magnitude and expected recurrence interval of storms which are capable of generating large increases in landsliding.

With the exception of earthflows, only limited applicable data exists describing the effect of annual precipitation, antecedent precipitation and storm magnitude (depth, duration and intensity) on the initiation of debris sliding. Some of the data has been evaluated for the

San Francisco Bay area (Ellen et. al., 1990), for the central Oregon Coast Ranges, and in other parts of the world, but little information is available for conditions comparable to the north coast. The public and land managers have historically attributed the occurrence of increased rates of debris sliding to individual storms, but the relationship is typically complicated by unique conditions of antecedent precipitation and specific storm characteristics. Often, storms of comparable "magnitude" produce markedly different results in a watershed. Because of a lack of data, storm characteristics for individual watersheds are generally unknown. When combined with the complicating influences of land management, the relationships attributed to each contributing mechanism may be indecipherable.

A useful study has recently been undertaken in the Elk River watershed to evaluate flood producing storm events in that drainage (Conroy, written communication, 1998). Similar research might be employed, using existing precipitation and runoff data in the lower Eel River area, to better define the magnitude of landslide producing storms.

25/4 Sediment deposits also exist along portions of the Bear Creek channel from aggradation events which predate management activities in the watershed. Sediment stored in the channel is not all management-related. We agree that the Bear Creek watershed has experienced cumulative watershed effects. However, fish densities, habitat conditions and riparian data suggest that significant recovery had occurred by the summer of 1996. As well documented in the Redwood Creek watershed, gravel bedded streams on the north coast, with characteristics similar to Bear Creek, can recover their original thalweg and profile elevation over periods of 10 to 30 years. In these examples, stored sediment remains in longer term storage compartments along the channel margin while the main thread of the active channel has recovered many of its pre-disturbance characteristics (Madej, 1995; Pitlick, 1995).

25/5 Dr. Reid, perhaps with good reason, was quick to embrace the hypothesis that large woody debris may have moderated downstream impacts in Bear Creek during the pre-management period, and that currently diminished quantities of large woody debris may have resulted in "substantially larger" impacts than would otherwise be expected. Although a logical argument, it was largely based on observations from our aerial reconnaissance work in Oregon and Washington following the 1996 storm event (PWA, 1996). Additional research and literature review, which we were unable to conduct for this project, would be needed to verify the importance of this process in Bear Creek.

26/2 We have already commented on problems with concluding that a cause-and-effect relationship exists between second-cycle logging and landslide rates after the 1997 storm in Bear Creek. Data from Jordan Creek does not support Dr. Reid's logging/landsliding relationship and conclusion, and such information casts serious doubt on the apparent causal relationship in Bear Creek. Additional research is warranted and necessary before the preliminary findings can be refined and extended outside of the watersheds where the data was originally collected. In addition, as incorrectly stated in the Bear Creek report (10/2), it now appears likely that storm(s) in the 1990s (specifically the 1997 storm) were of equal or greater magnitude than first-cycle storms (1955, 1964)(see discussion elsewhere).

27/2 We believe it would be unprofessional to ignore the treatment of road related erosion and to not treat these sediment sources simply because they accounted for only 8% of past sediment production in the basin. Road treatments can cost-effectively correct and prevent most road-related sediment sources that are "waiting" to fail and deliver sediment to the stream system, including surface erosion. We believe it is possible to cost-effectively prevent the delivery of over 55,000 yds^3 of future sediment to the stream channel system by the implementation of a variety of straight-forward erosion prevention treatments. The potential benefits are actually much larger, in that stream diversions and the resulting gully erosion will also be prevented. Because of inherent uncertainties in estimating future erosion, the potential reduction in future sediment yield from these processes was not included in the estimated sediment reduction calculations.

The US-EPA and NCRWQCB are currently developing TMDLs for most watersheds in northern California. The goal of this work is to determine what percentage of future erosion in the watershed is preventable. The three currently developed TMDLs for California all require an 80 to 90% reduction in road-related sediment yield.

Most road-related erosion can be identified before the erosion occurs and it can be prevented by pro-active implementation measures. In contrast, once forests are harvested (if harvesting is assumed to be the causative mechanism leading to slope failure) there is literally nothing that can be done to immediately remediate the conditions and prevent future slope failure. The hillslope is at the mercy of natural driving mechanisms (future storms).

In contrast to the treatment of road-related erosion, the best and most cost-effective techniques to minimize the effects of future silviculture on potentially unstable hillslopes is through the application of avoidance or harvest modification measures. Slopes which are sensitive to slope failure in Bear Creek occur most commonly on steep inner gorge areas, and secondarily on less steep streamside slopes and steep headwall swale areas. The scientific literature is replete with examples depicting the relatively high frequency of landslides in these geomorphically sensitive hillslope locations. We have identified these areas as sites in Bear Creek which merit close analysis by PL and have suggested the potential application of restrictive and protective land use practices. It is up to the Company and the responsible agencies to practice adaptive management to reduce the risk of future failures from these sites.

31/6 Clearly, it is neither prudent nor recommended to collect, concentrate and divert road surface runoff onto unstable or potentially unstable hillslopes. The object is to make the road as "hydrologically invisible" as is possible by dispersing runoff and not allowing it to concentrate. The best way to accomplish this is by local outsloping or by the addition of numerous ditch relief culverts and/or rolling dips. It is difficult to believe that Dr. Reid would encourage the use of long uninterrupted ditches to drain the entire roadbed and cutbank directly to stream crossing culverts. That practice was discontinued in California over 20 years ago with the implementation of the Forest Practice Rules. Although many land owners and agencies have been slow to implement the strategy of hydrologically disconnecting their older roads from the adjacent stream systems, efforts at inventorying and correcting these drainage problems are now underway on a number of ownerships.

37/1 By "natural association," we meant "natural geomorphic association." Taken in context with the remainder of the text, the meaning was clear and should not have been confusing..

37/2 In the Bear Creek report we have clearly identified the most common and obvious association with landslide occurrence to be steep inner gorge slopes. It is a physical, geomorphic association. These are also the slopes which are likely to be most sensitive to management, and the preliminary data suggests this is the case. We concur that it would be useful to perform a study that would "identify patterns of failure, for a variety of storm sizes, over a variety of site types, over a relatively large area," but the magnitude of that research effort would have been well beyond the scope of the Bear Creek assessment. We do not agree that field inspections by trained, experienced geologists, to identify suspect geomorphic, geologic, hydrologic and vegetative conditions, in the absence of such background information, would be "inadequate." Signs of physical instability need not be present for an experienced and qualified geologist to recognize conditions which are conducive to potential failure, and to evaluate risk on the hillslope as a result of a variety of silvicultural schemes.

As stated earlier, the MWAS does not significantly differ from current USFS strategies for recognizing, and avoiding or mitigating, potentially unstable terrain associated with the development of federal timber sales and subsequent timber harvesting in northern California. Perhaps Dr. Reid's point is that past regulatory review of timber harvesting plans has not considered avoidance, deferral or significant harvest modifications to be appropriate or acceptable mitigation techniques when addressing resource concerns on private property. If this is true, the past perception may have lead to a reluctance by inspecting geologists to recommend such mitigations in settings where they might otherwise be technically appropriate. Based on the recognized sensitivity of portions of the Bear Creek watershed described in the report, we do not believe that this perception still exists, or that there is (or should be) a reluctance by geologists to prescribe appropriate protective measures for hillslopes in the Bear Creek.

Finally, as a sidebar, the TMDL sediment source allocations currently being developed for the Garcia River, Redwood Creek and the South Fork Trinity River are recommending landowners reduce harvest-related landslide delivery anywhere from 40% to 60% over the next 30 years. We suggest, in the absence of any new, accurate analytical models to predict slope stability, the best available option is to rely on qualified, experienced geologists to conduct field inspections to assess the risk of management activities influencing slope failures. Armed with the proper assessment and implementation tools, equivalent reductions in long term yield should be feasible in Bear Creek.

37/3 Dr. Reid does not provide the calculations she uses to critique and contradict the statement in paragraph 37/3, so it is not possible to directly respond to her concerns. The statements in the paragraph, that landslide rates in Bear Creek are naturally high, are not intended to be misleading. They simply state what might not be obvious to someone who has not reviewed the pre-management aerial photography.

Dr. Reid's concluding statement that: "The bottom line is that the [Bear Creek] report demonstrates that present land use practices are directly responsible for at least a 960% increase in landslide frequency..." is without merit. It is a serious scientific mistake to interpret an apparent association between variables (such as recent harvesting and landsliding) as a "direct" cause and effect relationship without the data and proof to make such a claim. When proclaimed by a respected scientist, such interpretations quickly find their audience and work their way into debates about management and conservation, and into the regulatory arena for areas and practices which may have little in common with the original site. Without any new or original data, and without the benefit of the actual data collected for the Bear Creek study, it seems inappropriate to make such sweeping statements and conclusions.

Appendices: Note: The footnotes which are described by Dr. Reid at the end of her review are a part of the agency/PL agreement for the Interim Aquatic Strategy. They are not a part of the Bear Creek report except by reference.

1. We agree that a "broader understanding of the associations between landslides, site type, storm size and silvicultural practices, as a part of a broad scale analysis of landslide distribution," is needed and would be useful to have. Unfortunately, the science has not been done and such a multi-year scientific investigation is well beyond the scope of the Bear Creek project. Until such time as this effort is undertaken and completed, site specific analyses conducted by trained geologists will provide the best information available with which to identify and manage potentially unstable terrain. As stated earlier, this procedure is consistent with current "state-of-the-art" landslide risk analysis procedures being employed by the USFS.

Concluding comments

We appreciate the opportunity to address many of the review comments regarding the Bear Creek study and report. Dr. Reid's review was lengthy and required close analysis. A number of the comments provided by her were thoughtful and merited elaboration or explanation. We have developed our discussion of the review comments according to our assessment of the significance and relevance of the major topics presented, rather than presenting a point-by-point, repetitive discussion or addressing other relatively minor points or suggestions that were made.

We must be honest in stating that we were disappointed at the often hypercritical nature of Dr. Reid's comments. We have had occasion to work with her in the past and found her to be professional, creative and generally objective on other topics. For these reasons, some of her comments seemed out-of-character. Finally, we were also disappointed with the tone of a number the comments, where she appeared more interested in questioning our judgement, objectivity and professionalism than in the basic data and interpretations we presented. We have never been overly sensitive to constructive criticism or to recommended suggestions for improvements in our procedures or interpretations of data, as long as such criticism is itself unbiased and professional. For the last 20 years we have worked diligently with county, state, and federal agencies; Indian tribes; private landowners; industrial clients; and environmental groups to encourage a science-based approach to minimizing the erosion and sedimentation effects of common forest land management activities, and to identify and address the root causes of accelerated erosion which impact fisheries resources. We have developed a strong reputation from all quarters for professional, objective, science-based investigations, and we are very protective of that reputation.

In the Bear Creek study and report we have presented what we hope is a significant step forward in our understanding of watershed processes. It is one contribution to the body of knowledge and we encourage landowners, scientists and regulatory agencies to look at the data and to review the findings to see how it relates to their knowledge and base of experience. We have prepared this response to the review comments largely to set the record straight regarding the procedures we employed, the assumptions we made, and the interpretations we report in the final document. As with any such study, new information continues to become available after the report has been prepared and distributed. We have included some of this clarifying information (for example, our increased understanding of the magnitude of the storms of the 1990s and the contrasting landslide data for the adjacent Jordan Creek watershed). As new information is developed, original interpretations must be adapted and modified to best account for the data. As a limiting and controlling factor in conducting the Bear Creek study (as with any comparable undertaking), the short time frame available for data collection, analysis and reporting will always play a role in determining what data can feasibley be collected and what analyses can then be undertaken.

We believe the Bear Creek report represents a significant new database which describes the complex interaction between natural watershed processes and land use activities. We have attempted to provide objective information in the report which will allow other professionals, agencies and the general public to evaluate the information presented and to develop their own opinions and interpretations of the data. However, we strongly encourage readers of the report, Dr. Reid's review comments, as well as this response, to be very cautious about applying the data or the preliminary interpretations (either ours or theirs) from this small drainage to other watersheds in northern California, no matter how similar they may appear or how close they may be. As Dr. Reid suggested in her review, a much broader based investigation would be required in order to understand the complex and true nature of the relationship between sediment production and land use practices.

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Attachment II-2

Dr. Dean Thompson's Environmental Risk Assessment of Herbicide Use on Forest Lands of the Pacific Lumber Company, Scotia, California

Environmental Risk Assessment of Herbicide Use on Forest Lands of the Pacific Lumber Company, Scotia California

by

Dean G. Thompson (Ph.D.)

Final Report

submitted to:

Pacific Lumber Company, Scotia California

November 13, 1998

Environmental Risk Assessment of Herbicide Use on Forest Lands of the Pacific Lumber Company, Scotia California

Table of Contents

1.0 Preface

The following document is an assessment of the environmental chemistry, fate and toxicology of several forest-use herbicides pertinent to key issues raised in the Environmental Impact Statement (EIS) (Authors, date).

In developing this assessment a two-day site visit to the land holdings of the Pacific Lumber Company in the vicinity of Scotia, California which comprise the area of concern (AOC) was undertaken. During this period, interviews and discussions were held with several key company representatives including Dr. D. Opalach, Mr. S. Chinnici and Mr. M. Rodgers, with subsequent follow-up communications with Mr. T. Koler, who supplied information pertaining to overall forest management and site characteristics, biological species of concern, herbicide use patterns and soil characteristics respectively. The author takes this opportunity to thank these individuals for the time, courtesy and professional conduct and input. As a result, the author has become familiar with the materials, techniques and management objectives of PALCO's vegetation control program and use of herbicides, which in my professional judgement are consistent with federal and state legislation, product labels, as well as standard forestry practices.

Within the context of standard practices for herbicide use within the AOC, an extensive review of the scientific literature pertinent to key issues was conducted and used as a basis for assessing risk to amphibian, reptilian, piscine and avian species of special concern (red-legged, yellow-legged and tailed frogs northwestern pond turtle, coho salmon yellow warbler and yellow-breasted chat). The assessment is based upon principals of environmental chemistry, toxicology and risk estimation principals and takes a "weight of evidence" approach.

In conducting any risk assessment a significant amount of professional judgement is required. As a professional research scientist with more than 12 years experience and 40 scientific journal publications in environmental chemistry and toxicology (see attached curriculum vitae), I feel qualified in applying this judgement. While every attempt has been made to ensure the accuracy and objectivity of the review, the author assumes no legal liability for inaccuracies or oversights which may have inadvertently occurred. The assessment was conducted solely on personal time and represents the personal viewpoints of the author. Any mention of product tradenames are for information and clarity purposes only and do not in any way imply product endorsement. In all cases pesticides must be handled and applied according to specifications on the product label which may overrides or supercede any recommendations made here in.

Environmental Risk Assessment of Herbicide Use on Forest Lands of the Pacific Lumber Company, Scotia California

2.0 Executive Summary

The following report summarises a site-specific environmental assessment pertaining to the use of forest-vegetation management herbicides on private land holdings of the Pacific Lumber Company in the environs of Scotia, California which hereafter will be referred to as the area of concern (AOC). Seven different herbicide products (atrazine, sulfometuron-methyl, glyphosate, hexazinone, triclopyr, 2,4-D and imazapyr) are currently or may be utilised for control of competing vegetation, roadside weeds and invasive exotic species such as pampas grass (*Cortaderia selloana*). All of the herbicide products used within the AOC are applied strictly according to label specifications and are registered for such uses by both federal (Environmental Protection Agency (EPA)) and state (California Department of Pesticide Regulation (CDPR)) agencies. This fact, in and of itself, strongly supports the concept that significant deleterious effects in the general environment or to common animal species are not to be expected. However, both abiotic and biotic variables peculiar to the AOC may generate unique environmental concerns, thus warranting a site-specific review. In this regard key issues which have been identified relate to:

Key Issues

a) the potential for herbicides to move off-site and contaminate aquatic environments
b) risks to amphibian species of special concern (red-legged, yellow-legged and tailed frogs, southern torrent salamanders)
c) risk to reptilian species of special concern (northwestern pond turtle)
d) risk to piscine species of special concern (coho salmon)
e) risk to avian species of special concern (e.g. yellow warbler, yellow-breasted chat)

To assess risk potentials associated with these issues, a site-visit was conducted to examine and acquire information on key factors pertaining to overall forest management, site characteristics (e.g. topography, climate, soils, hydrology), biological species of concern, and to view standard practices associated with herbicide use. Subsequently, a review of the pertinent scientific literature on the environmental chemistry, fate and toxicology of these herbicides was undertaken. The potential for herbicides to move off-site was determined through examination of their key physicochemical characteristics (e.g. water solubility, partition coefficients for organic carbon and soils) and estimation of environmental persistence and fate from relevant laboratory and field studies. A reasonable worst-case scenario approach was taken in assessing toxicological risk potentials wherein maximal concentrations expected in the environment were compared to minimum median lethal concentrations as reported in the literature for the most sensitive avian, fish and amphibian species. While relatively crude, subject assumption and extrapolation errors, and arguably

hyperconservative, such quotient methods are simple and widely used as a first step in risk assessment (Rodier and Mauriello, 1993; Finley and Paustenbach 1994; Environment Canada, 1996). As noted by Suter et al. (1992), risk assessment differs from risk analysis in that the latter involves an explicit, quantitative consideration of uncertainties and the expression of the final estimated effect as a probability. Uncertainties may enter estimation of ecological risk from several sources and include for example, intraspecies variability (Blanck 1984, Hughes et al. 1989), interspecies variability and laboratory to field extrapolation. The limited ecological relevance of laboratory studies in combination with uncertainty components as noted above, preclude direct statements of risk based on quotients of estimated environmental concentrations (EEC) to laboratory derived toxicity endpoints (e.g. LC_{50}). Therefore, a margin of safety approach has been employed wherein in quotients less than 2 are ranked as relatively higher risk, quotients from 2-10 are ranked as moderate risk, and 10-100 are ranked as low risk. Since herbicides considered here are rapidly dissipated or degraded in the natural environments, chronic exposures are not anticipated and the risk assessment is focused largely on potential for direct, acute effects.

Review of the scientific literature suggests that a substantial database on environmental chemistry, fate and toxicology of these herbicides is available from laboratory studies and sufficient to support general environmental risk assessment for the species of concern in most cases. Further the site and natural history of key species of concern to which the assessment pertains are reasonably well characterised. Finally, laboratory data is significantly augmented by the results of field studies conducted in forested ecosystems throughout North America which have focused on both fate and effects of these herbicides under more realistic environmental scenarios. As most field studies are conducted under reasonable worst-case conditions of direct overspray, results may be considered as providing approximate upper bounds for expected effects. While databases on fate and impact of the classical and most frequently used herbicides (atrazine, glyphosate, 2,4-D, triclopyr, hexazinone) are most comprehensive, those for more recently registered compounds (e.g. sulfometuron-methyl, imazapyr) are still developing. Toxicological information on avian and fish species is relatively extensive, less comprehensive data is available for amphibians and pertinent data for reptiles and amphibians is largely unavailable.

All herbicides currently used or expected to be used within the AOC are moderately to highly water soluble and non-bioaccumulatory. They are subject to a variety of abiotic and biotic degradation mechanisms as well as several dissipation pathways, and are therefore generally non-persistent in plant, soil, or aquatic compartments of the environment. Principal degradative metabolites of all herbicides have been identified and are generally more water soluble, more susceptible to metabolism and less persistent than their respective parent compounds.

a) the potential for herbicides to move off-site and contaminate aquatic environments

Comparative physico-chemical properties, as well as results of both laboratory and field studies (**Risk Matrix 1**) demonstrate that neither glyphosate nor triclopyr pose a significant risk for contamination of streams via leaching or runoff. If it were to be used within the AOC, 2,4-D might pose a moderate risk of stream. However, owing to its rapid degradation in soils, 2,4-D is not considered susceptible to leaching and would only be mobililized to any significant extent with surface runoff where rainfall events occur shortly after application. Application of ester formulations of 2,4-D and triclopyr which have significantly higher binding affinity for organic carbon, is an important factor further mitigating against stream contamination potentials for these compounds. The risk of stream contamination by imazapyr is assessed as moderate, largely on the basis of limited or equivocal field study results. Although its physicochemical properties suggest a potential for leaching, field results from two different forestry studies fail to corroborate this. Similarly, field studies have shown conflicting results with regard to potential for surface runoff. Potential risk associated with this herbicide is obviated by the fact that it is not currently used within the AOC. Decisions taken to use this product should be made under an adaptive management and monitoring approach as described below. The greatest risk of stream contamination is associated with the triazine herbicides, atrazine and hexazinone. These compounds have physicochemical properties conferring leaching and runoff potential and for which several field studies conducted in forest environments document either leaching below 30 cm soil depth or runoff with surface water. The fact that hexazinone is not currently used within the AOC negates any potential for stream contamination by this compound and any decision to use this product must be taken with due consideration for its potential to contaminate aquatic environments.

The potential for herbicides to move off-site via leaching or surface movement is a complex function of several variables including rate and method of application, timing of herbicide application relative to periods of highest or most intense rainfall, general soil characteristics and degree of moisture content at time of application, slope, surface and subsurface channelling, degree of vegetative or sorptive material remaining on the site and existence of vegetative buffers surrounding drainage streams. Conditions within the AOC which enhance the risk of offsite movement for susceptible herbicides such as atrazine, include slope (> 50% slopes in more than 50% of each of six constituent watersheds), periodic high rainfall (particularly January through March), and the common occurrence of ephemeral channels on treated sites. Factors which mitigate against risk of offsite movement include; targeted application, vegetation, large woody debris and organic carbon resulting from prescribed burning which remain on the site. Management practices which require retention of vegetated riparian 170 and 100 foot buffers about all class I and II streams respectively, are particularly important mitigation mechanisms. Given the relatively high organic matter content, cation-exchange-capacity and fine-texture of soils within the AOC, movement of atrazine via leaching would not be expected to be significant. However, the use pattern for this herbicide, which typically involves applications during January, February, April and May when rainfall frequency and intensity are greatest and the fact that it may be applied to or near ephemeral stream channels may increase the risk of runoff losses. In 1998, the company began a replacement program whereby sulfometuron methyl was applied in place of atrazine. Based on the analysis presented in Risk Matrix 1, and considering its substantially lower use rates as compared to atrazine, sulfometuron methyl is not expected to contaminate streams. This postulate is supported by the studies of Wauchope et al. (1990), Michael et al (1987), Lym and Swenson (1991) and Wehtje et al. (1987) which all provide evidence that under environmental conditions pertinent to this assessment, sulfometuron methyl has little potential for movement in soil water. Several monitoring studies conducted collaboratively with the North Coast Regional Water Quality Control Board (NCRWQCB) at stream sites proximal to treatment areas have specifically addressed risks of stream contamination. These studies have repeatedly found non-detectable residues of glyphosate, triclopyr, sulfometuron-methyl, atrazine and diesel fuel under both baseflow and stormflow conditions in Class I and II streams. These results provide strong evidence that the combination of best management practices currently employed within the AOC, including the restriction for ground applications only, employment of 170 and 100 foot buffer zones about all Class I and II streams respectively, incorporation of tracer dyes to enhance visualization during and after applications, supervision of herbicide applications by trained staff, effective site mapping and on-site control of contract applicators, mitigate the risk of contamination of Class I and II streamwaters to insignificance. No monitoring information documenting residues in ephemeral stream channels or in sediments were available.

b) ***risks to amphibian species of special concern (yellow-legged frogs, red-legged frogs, tailed frog, southern torrent salamanders)***

Characterization of the exposure risk for the various amphibian species of special concern within the AOC is species and behaviour dependent. Since yellow-legged frogs are seldom found far from small, permanent streams, direct exposures would be predicated on contamination of Class I and II stream systems. The results of several monitoring studies provide strong and direct evidence supporting the contention that yellow-legged frogs are not exposed to significant levels of any herbicide currently used within the AOC. This includes atrazine for which the risk potential is considered greatest . Further, results reported in the literature for field studies involving direct overspray scenarios show maximal residues which are substantially lower than median lethal toxicity values for frogs (**Risk Matrix 2**). Finally, these data indicate that the toxicological risk potential is greatest for those herbicides (glyphosate and triclopyr) which have the least potential to contaminate streams.

For tailed frogs, which are highly specialized to live in swift, perennial streams with low temperatures (Nussbaum et al. 1983), the risk of toxicological effect is also considered minimal. Again, given that even maximal residues of forest herbicides are substantially below median lethal toxicity values for various frog species, the risk of direct lethality to tailed frogs within the AOC where herbicide concentrations are below detection limits is exceedingly low. In this case, the fast-flowing stream habitat preferred by this species, provides an additional mitigating factor since several studies show that durations of exposure in such systems (1-2 h) are significantly shorter than typical laboratory exposure periods (48-96 hr).

Treatment sites, which are typically recently cutover and burned, represent extremely marginal habitat for red-legged frogs and southern torrent salamanders, which prefer cool, moist, areas of late seral stage forests. However, anecdotal observations of red-legged frogs on herbicide treatment sites indicates that such exposures are not impossible. Similarly, it is possible that even within cutover sites, southern torrent salamanders may inhabit seeps or springs containing flowing water. Since these salamanders are rarely found more than one meter from water (Anderson 1968, Nussbaum and Tait 1977), avoiding overspray of seeps and springs should prevent direct exposures to salamanders. Habitat preference, low herbicide application frequency (2 out of every 60 years), low total percentage of the landbase treated annually and the fact that treatment areas are non-contiguous are all elements inherently mitigating against the risk of direct exposures to red-legged frogs and southern torrent salamanders within the AOC. Also, since herbicides used within the AOC are applied by ground application techniques typically involving teams of backpack sprayers and since biota have a natural tendency to avoid humans, these amphibians would be expected to move away from applicator teams and seek cover as they pass through the area, further reducing the potential for direct exposure. Given the vagaries of potential exposures, the lack of toxicological data relevant to cutaneous exposures for adult frogs and the complete lack of toxicological information on toxicity of herbicides to salamanders it is not possible to directly assess risk to these species. However, amphibians as a group do not appear to be unusually sensitive to herbicide intoxication.

c) risk to reptilian species of special concern (northwestern pond turtle)

The specific habitat of the northwestern pond turtle includes includes drainage ditches (Zeiner et al. 1990a, Bury 1962, Holland 1994, and Nussbaum et al. 1983). In addition, this species is known to move hundreds of meters away from aquatic areas and nest in terrestrial sites. Thus, there is a potential for these turtles to be directly exposed to herbicides either on terrestrial treatment sites or when herbicides are applied around ditches for roadside weed control. No toxicological information could be found upon which risk to reptiles could be evaluated.

d) risk to piscine species of special concern (Coho salmon)

Coho salmon migrate into freshwater tributaries in late fall and winter and lay eggs in swift flowing gravel-bottomed streams. Alevins and young typically inhabit such areas for up to a year following spawning. A comparison of literature values for maximal concentrations of herbicides observed in forest field studies with median lethal concentrations for the most sensitive freshwater fish species **(Risk Matrix 2)**, suggests that only triclopyr and 2,4-D esters present a significant risk to coho salmon. Since these herbicides are applied by ground application only to treatment areas outside the 170 and 100 foot buffer zones protecting Class I and II streams, potential direct contamination of streamwater is considered negligible. Field and laboratory studies (Bentson and Norris,1991; Thompson et al. 1994) demonstrate that ester forms of these compounds are rapidly hydrolzyed or photochemically degraded to the acid form upon release to the environment, indicating that triclopyr residues entering streams by runoff or leaching mechanisms would be in the least toxic acid form. Several studies have documented that realistic worst-case exposures regimes for triclopyr ester in flowing waters are of insufficient magnitude and duration to generate siginificant toxic effects in salmonid fish species (Thompson et al. 1991; Kreutzweiser et al.1995). In the case of coho salmon the low potential for sublethal or indirect effects is supported by several studies on the most toxic compound (triclopyr ester) which show no effects on aquatic macroinvertebrates comprising the majority of their diet (e.g. Mayes *et al.* 1989. Fontaine 1990; Kreutzweiser et al. 1995) and by the studies of Janz et al. (1991) and Kreutzweiser et al.(1995) which demonstrate no significant physiological stress or growth effects in sensitive salmonid species even following direct exposures. Given that several monitoring studies show non-detectable residues of triclopyr in streams within the AOC, there appears to be essentially no risk of either direct or indirect toxicological effects of either triclopyr or other herbicides on coho salmon.

e) risk to avian species of special concern (yellow warbler, yellow-breasted chat)

Since herbicides used within the AOC are applied by ground application techniques only, to only a very small percentage of the total landbase annually and that they are applied to non-contiguous regeneration sites on a temporal frequency of typically 2 out every 60 years, direct exposures to avian species within these systems must be characterized as improbable and infrequent. This is particularly true for the yellow warbler and yellow-breasted chat which nest in riparian habitats. These birds would be expected to make minimal use of herbicide treatment sites and would be expected to move away from applicator teams and seek cover as they pass through the area, making direct exposures unlikely. Based on worst-case exposure **(Risk Matrix 2)**, indirect exposures resulting from herbicide contamination of insects and berries which form the bulk of their food is not considered sufficient to induce toxicological effects in most cases. As exceptions, the risk matrix suggests that atrazine and 2,4-D have relatively high potential for effects on birds. However, for these species indirect effects through habitat alteration or food reduction are also unlikely since herbicides are not used directly in their primary foraging habitat. Several studies (MacKinnon and Freedman 1993; Slagsvold 1977; Morrison and Meslow 1984a&b; Santillo et al 1989; Hardy and Desgranges 1990) fail to demonstrate substantive population reductions in neotropical migrant birds during the growing season after treatment and show that changes in utilization of herbicide-treated sites is specific. Where species densities are reduced, they recover during the following growing season. Since these studies involve direct over spray of regenerating clear-cuts in which those species were nesting and foraging, it is unlikely that indirect exposures as would occur within the AOC would have substantial effects on yellow warblers or yellow-breasted chats.

Considering the ground-based methods and low frequency of herbicide application to any given site within the AOC, the non-persistent nature of the compounds, their general lack of tendency to move off-site (exception atrazine and hexazinone) and the existence of 170 and 100 foot untreated riparian buffers around all class I and II streams including seeps and springs, the overall risk of direct toxicological effects to amphibians (yellow-legged frogs, tailed frogs, southern torrent salamanders) and fish (coho salmon) appears to be insignificant. This postulate is strongly supported by the fact that on-site monitoring studies have failed to show detectable residues for any herbicide in streams which are the preferred habitat for these species. The postulate is further supported by scientific literature which indicates a greater than 2 fold margin of safety between maximally observed streamwater concentrations and laboratory median lethal toxicity estimates for most herbicide-species combinations. Highest toxicological risks appear to be associated with the esters of 2,4-D and triclopyr and for ROUNDUP formulations which have the least potential to mobilize and contaminate these stream systems via leaching or surface runoff. The existence of 100 to 170 buffers vegetated buffers about these systems is a significant mitigating factor. The fact that no herbicides are employed within these buffer zones is protective not only of the aquatic habitat but also of species such as the yellow warbler and yellow-breasted chat for which riparian zones are preferred habitat. Under the conditions of herbicide use within the AOC, there appears to be no scientific basis upon which a signficant risk to these bird species may be postulated. Of the key issues outlined for this assessment the greatest risk potentials may be for red-legged frogs and northwestern pond turtles. Although the potential for significant effects on these species also appear to be minimal, insufficient toxicological and exposure information results in an equivocal assessment of risk.

Based on this site-specific assessment and evaluation of the pertinent scientific literature, the risk of offsite movement of most herbicides and potential toxicological effects on most species of concern were considered to be mitigated to insignficance. As such current best management practices were considered satisfactory and were recommended to be continued. A modified monitoring is proposed to provide further information pertaining to the minimal, equivocal, risks which may be associated with potential runoff losses in ephemeral stream or potential herbicide effects on red-legged frogs and northwestern pond turtles. The recommended monitoring program is designed to focus on maximal runoff potentials associated with the most mobile compounds during the first two storm events following application as well as aqueous and sediment residues in critical habitats for red-legged frog and northwestern pond turtle species of concern.

3.0 Overview and Scope of Assessment

The following environmental risk assessment was conducted in response to concerns raised in the Environmental Impact Statement (EIS) pertaining to the possible sale of that section of PALCO lands known as the Headwaters area. The scope of this assessment is restricted specifically to the use of herbicides in on private forest land holdings of the Pacific Lumber Company (PALCO) in the environs of Scotia California which hereafter will be referred to as the area of concern (AOC). The assessment considers only those herbicides and use patterns which are currently or which may be employed within the AOC. Even within these constraints, comprehensive discussion of all aspects of herbicide chemistry, fate and toxicology is beyond the scope of this document and readers seeking further detailed information are referred to a variety of texts on fundamental aspects pertinent to the subject (McEwen and Stephenson, 1984; Worthing and Hance 1991; Cheng 1990; Howard, 1991; Grover and Cessna1991;Rand and Petrocelli, 1985). Previous reviews (Ghassemi *et al.* 1984; Sacher 1978; Tooby 1985; NRCC 1978; Stewart 1991; WSSA, 1983; Reinart and Rogers 1987; Veith *et al.* 1989, Brandt 1984; Hunter 1984; Solomon et al. 1996) document the environmental fate and effects of herbicides generally. Reviews that focus on specific aspects of the environmental chemistry and toxicology of herbicides in relation to forestry uses include Roshon et al. (1998), Neary (1993), Lautenschlager (1993), U.S.D.A.-F.S. (1984), Norris 1981; Freedman (1991).

This assessment is focused on key issues as identified in the EIS including:

Key Issues

a) the potential for herbicides to move off-site and contaminate aquatic environments
b) risks to amphibian (southern torrent salamander, red-legged, yellow-legged and tailed frogs) and reptile species (northwestern pond turtle) of special concern
c) risk to piscine species, particularly coho salmon and
d) risk to avian species of special concern (e.g. yellow warbler, yellow-breasted chat)

All of the herbicides (atrazine, glyphosate, hexazinone, 2,4-D, sulfometuron-methyl, imazapyr) used within the AOC and considered in this review are registered both federally (EPA) and within the state of California (California Department of Pesticide Regulation). As such there is a considerable body of scientific evidence substantiating the fundamental safety and environmental acceptability of these compounds when used as prescribed on the product label. A component of the regulatory database arises from laboratory or field studies which may not be pertinent to the species of special concern or conditions extant within the AOC. Therefore, this assessment is focussed on abiotic and biotic variables associated with the AOC which may impinge on extrapolative assessment of risk from the available scientific database. In the risk analysis portion of the document a reasonable worse-case scenario approach has been taken in an attempt to identify any risks which are substantively supported by the available science. Throughout the assessment judgements are based on the weight of evidence principal. In cases where any potential risks are identified actions and adaptive management strategies are recommended which should mitigate the risk to insignificance.

4.0 Site Description

4.1 General Location and Description of PALCO Lands Comprising the Area of Concern

Forest lands of the Pacific Lumber Company (PALCO) considered in this environmental assessment are located near Scotia, California in Humboldt County and will be generally referred to hereafter as the area of concern (AOC). The AOC covers portions of six distinct watersheds as delineated by the state of California, namely Humboldt Bay, Mad River, Yager, Van Duzen, Eel, and Bear-Mattole, within which PALCO ownership comprises 202,091 acres or 24% of the combined land area of these watersheds (FWEC, 1998). General characteristics of climate, geomorphology, topography, hydrology and biology for the AOC are effectively summarized in the environmental impact statement (EIS). As such, only those aspects which critically affect the environmental fate and impact of herbicides on these lands will be considered in this assessment Consistent with standard industrial forest practice, herbicides are applied on the majority of the cut over landbase to maximize reforestation and growth of the coniferous crop. As a typical example, herbicide applications for 1997, herbicides were applied to 4,853 acres or about 2.4 % of total land area in the AOC.

4.2 Climatic Influences

Principal climatic factors influencing the fate and behaviour of herbicides in the environment are temperature, precipitation, and sunlight irradiation which control key degradation (hydrolytic, photolytic, biological) processes and which may indirectly or directly influence dissipation processes (sorption, advection, volatilization). Climate of the AOC is typical of northern California characterized by moderate temperatures (40-70 °F), summer fogs, and annual precipitation of 43 to 80 inches annually. Low temperatures may impair herbicide dissipation and degradation through inhibition of volatilization, microbiological and hydrolytic degradation mechanisms, while higher temperatures tend to accelerate all of these process. Mean maximal temperatures (62.9 °F; min 54.8 °F in January; max 70.6 °F in September) as recorded at Scotia, reflect generally moderate temperature conditions and little seasonal variation typical of marine coastal environments. Such temperature regimes are well within the range for optimal microbiological activity and thus microbial degradation of herbicides in this area would not be expected to be temperature-limited at any time of the season. While physical-chemical reactions associated with volatilization, and hydrolysis generally increase directly with higher temperature, temperatures characterizing the AOC are sufficient to ensure that these processes will occur and will not be unduly inhibited at any time of the year.

Effects of precipitation on herbicide fate and behaviour vary with type, amount, intensity and relative timing of application and precipitation events. Given the moderate maritime climate and elevations characterizing the AOC, snowfall is insufficient to warrant consideration. However, rainfall and fog represent key elements of the regional climate with the potential to significantly influence the fate and behaviour of herbicide residues. Annual occurrence of fog ranges from 40 to 70 days per annum along the coast of Humboldt County (Hardwick, 1973), with a clear seasonal pattern of increased frequency in summer and early fall during months of July through October (FWEC, 1998). Fog occurs principally in the lower river valleys and is less important at elevations above 600feet. Average annual precipitation in the six watersheds comprising the AOC are similar, typically ranging from 60 inches per year in upper reaches to 40 inches per year in lower reaches

(EIS 1998). Maximal rainfall occurs in the upper Eel River watershed (110 in/yr annually) and lower Bear-Mattole watershed (90 in/yr annually). Annual precipitation patterns are a key factor controlling soil moisture and thus microbiological activity and microbial degradation of herbicides. While the occurrence of summer fog in low-lying areas would ensure optimal soil moisture and microbiological activity year-round, higher elevations may experience summer droughts during months of May through September when average monthly rainfall is minimal (< 1 in./month). Low average rainfall during these months is typically insufficient to induce waterflow in ephemeral or Class III channels, thus essentially eliminating the risk of any transport of herbicide residues with surface water. Additionally, low rainfall frequency and maximal sorptivity of soils during the period would mitigate effectively against any potential for downward percolation or leaching of herbicides through the soil column. In contrast, the higher average rainfall levels and occurrence of significant storm events during winter months (October through April) induce greater risks of surficial movement and leaching.

4.3 Geology, Topography and Soils

A full review of the geological, topographical and soil characteristics of the AOC are beyond the scope and needs of this assessment. However, pertinent aspects of geology, topography and soils as they relate to herbicide environmental fate will be discussed herein. The AOC is typical of northern coastal California with substantial topographical relief (0 to 6,000 feet above sea level) associated with river valley bottoms and ridges. The AOC is comprised of three principal rock complexes, namely Franciscan Central, Franciscan Coastal and the Yager formation. The Franciscan central belt is characterized by melanges or large blocks of mixed rock including, in this case, conglomerates, sandstone, chert, limestone, metamorphic and igneous rocks originating in marine environments. Coastal Franciscan belt rocks consist mostly of broken marine sandstone without exotic melanges. A younger sequence of sedimentary rocks, referred to as the Wildcat group, rests on the Franciscan and Yager strata and are comprised predominantly of marine sandstone, mudstone and siltstone with minor amounts of river-deposited sandstone (Moley, 1992). The younger sedimentary (sandstone and siltstone) rocks of the Wildcat group are subject to weathering and generate the deeply dissected terrain with small steep tributaries which is characteristic of the region.

Soils of the AOC are mostly of the Larabee, Hugo, Hely, Atwell and Melbourne soil series (CDF, 1975). These soils range from 30 to 70 inches in depth and are loam or clay loam in texture with the exception of Hely soils which are sandy loams. Fine textured, deep soils such as these, particularly when overlain by forest litter would not be considered highly susceptible to leaching. Of the soils within the AOC, the Hely series is characterized by coarsest texture and thus maybe relatively more susceptible to leaching.

Since the adsorption-desorption process mediates both microbial degradation and mobility with soil water (either downward percolation - leaching, or lateral movement with surface water), the characteristics of the soil environment have a direct and significant effect on the ultimate fate of xenobiotics in the terrestrial compartment. Owing to their relative abundance and surface properties, clays and organic matter are the two most important constituents controlling adsorption/desorption. Clay content of soils differs according to its parent mineralogical nature as shown in Table 1.

Table 1. CEC and Surface Areas of Various Soil Components.

Soil	CEC (mEq/100 g)	Surface Area (m^2/g)
Organic matter	200-400	500-800
Vermiculite	100-150	600-800
Montmorilonite Clay	80-150	600-800
Illite Clay	10-40	65-80
Kaolinite Clay	3-15	7-30
Oxides and hydroxides	2-6	100-800

In agricultural and other "artificial" soil types, organic matter is typically low (0-10%) and adsorption/desorption is influenced more by soils texture (% sand, silt, clay). However, in forest soils and particularly in lowland sites, organic matter (depth and composition of the humus layer) are significantly greater and thus an important factor controlling adsorption. Three factors - soil texture, organic matter content and cation exchange capacity (CEC) are key soil characteristics controlling sorption and fate in the soil compartment. Generally speaking, the soils with higher OM, CEC and silt% will be more sorptive, reducing the potential for a given xenobiotic to move with soil water. The leaching of xenobiotics in soils is a major environmental concern because of potential contamination of groundwater and drinking water supplies. Several techniques are used to measure leaching and several physical characteristics of the pesticides however a relatively new approach has been developed (Gustafsson 1988) which makes use of two relatively easily measured parameters, the K_{OC} (adsorption coefficient on organic carbon) and the half-life in soil. These two numbers can be used to calculate the groundwater ubiquity score (GUS), according to the following formula:

$$GUS = \log DT_{50} \times (4 - \log K_{oc})$$

This formula has been tested in a number of field situations where pesticides have been found in groundwater. In California soils a plot of Log half-life vs Log (K_{OC}) allowed those pesticides that leached to be separated from those that did not. Using this method, the author suggested that compounds with GUS values less than 1.8 were not susceptible to leaching, while those with GUS values greater than 2.3 had a propensity to leach. GUS estimates for the herbicides under consideration here are provided in risk matrix 1, and suggests that glyphosate and sulfometuron methyl are least susceptible to leaching while hexazinone and imazapyr are most susceptible. It must be noted here that lab-based measures of leaching tendency are indicators only and are most appropriate for relative ranking of the potential for a herbicide to leach. Depending upon conditions within natural environments, degradation and dissipation processes may be either faster or slower than those estimated in the laboratory. Similarly, sorptive properties of soils may vary dramatically depending on moisture content, CEC and textures. Finally, in the forest context adsorption to surface litter is often a significant factor mitigating against leaching while the occurrence of large root channels may significantly enhance the potential for mass transfer to lower mineral horizons. For all of these reasons, assessment of leaching and runoff potential must consider parameters such as GUS which provide a relative indication of potential, laboratory studies such as soil column leaching and field study results in an overall weight of evidence approach.

4.4 Biotic Constituents of the Area of Concern

Given their known occurrence within the AOC, status as species of special concern in the state of California and potential exposure to herbicide residues used in forest vegetation management activities, several species of amphibians, fish, reptiles and birds are considered biotic elements requiring special consideration in this assessment. In this regard, key biota include the foot hill yellow-legged frog (*Rana boylei*) the red-legged frog (*Rana aurora*), the tailed frog (*Ascaphus truei*) and southern torrent salamander (*Rhyacotriton variegatus*); coho salmon (*Onchorhynchus kisutch*), the northwestern pond turtle *(Clemmys marmorata marmorata*),and two species of neotropical birds the yellow warbler (*Dendroicha petechia brewsteri* and *D.p. sonorana*) and the yellow-breasted chat (*Ictera virens*).

Although the yellow-legged frog is able to utilize a variety of habitat types(Zeiner et al. 1990a) it is seldom found far from small, permanent streams with banks that can provide sunning sites (Nussbaum et al. 1983, and Zweifel 1968). Localized declines in populations of this species are often attributed to habitat alteration, as well as predation and competition by introduced bullfrogs. Breeding and egg-laying usually occurs over two week periods from March to May, depending on water conditions and following flood water recession (Zeiner et al. 1990a). Eggs hatch in about five days with tadpoles transforming in two to three months. Important aquatic micro habitats include shallow, low velocity areas near gravel bars, and ground cover on stream banks (Kupferberg 1996, Lind et al. 1992). Substrate for egg laying and tadpole hiding cover is provided by interstitial spaces between cobbles and boulders in low gradient large stream segments, and large woody debris in side pools and channels (Kupferberg 1996, Lind et al. 1992). Avoiding disturbance of terrestrial habitat within approximately five meters of stream banks is beneficial in supporting important habitat features for this species (Kupferberg 1996).

Two distinct subspecies of the red-logged frog (*R. a. aurora* and *R. a. draytonii*)as well as a population exhibiting intergrade characteristics (in the area between southern Del Norte County and northern Marin County) are known to occur in northern California and are variously listed as threatened or as species of concern. While the intergrade frog seems relatively common and widespread, populations of the *R. a. draytoni* subspecies of the inland valleys have probably been in decline since the turn of the century due to commercial exploitation (Jennings and Hayes 1985). Specific habitat for red-legged frogs includes ponds, slow moving creeks, puddles, and drainage ditches in or near moist forests and riparian habitats (Nussbaum et al. 1983, Bury and Corn 1988b). However, during wet weather, individuals may disperse considerable distances from breeding sites (Zeiner et al. 1990a). The red-legged frog breeds from March to July laying eggs in shady, deep, pools with little or no flow and submerged vegetation (Nussbaum et al. 1983; Cockran and Thoms

1996; Hayes and Jennings 1988). Dense vegetation close to the water level and undercut banks appear to be essential for shade and protective cover (Nussbaum et al. 1983). Larval development takes from 11 to 20 weeks.

The tailed frog generally occurs in areas which receive over 40" of rain annually including Humboldt county (Bury 1968). Tailed frogs are highly specialized to live in swift, perennial streams with low temperatures (Nussbaum et al. 1983) and avoid slow-flowing streams or wetlands (Daugherty and Sheldon 1982). Although habitat for tailed frogs has primarily been found in mature and late seral stage coniferous forests (Bury 1983, Bury and Corn 1988, Welsh 1990, Welsh et al 1993), observation of individuals in younger forests suggests that factors other than forest age are important habitat determinants (Welsh 1990). Other habitat features of this species mentioned in the literature include > 85% canopy closure or < 22°C air temperature; < 14°C soil temperature, and > 40% relative humidity (Welsh et al. 1993, Bury and Corn 1989, Chen et al. 1993). An average 60m riparian buffer width has been suggested to maintain suitable air and soil temperatures, and relative humidity regimes (Welsh et al. 1993, Ledwith 1996). The adults may be abroad from April to early September, but this timing can vary with locality (Stebbins 1966). Tailed frog breeding generally occurs during the late summer to early fall (August-September), but pairs of frogs have been found clasped together at any time of the year (Nussbaum et al. 1983). Eggs are laid in rosary-like strings under rocks (Stebbins 1966), from late June to early August, with hatching in August and September. The tadpoles have suction like mouths, which allows attachment to rocks. Cobble and boulder substrates with relatively low embeddedness are important for the larvae (Hawkins et al. 1988, Altig and Brodie 1972). The larvae require 2-3 years to transform. The presence of woody debris in streams may be beneficial, or perhaps necessary, for micro habitat requirements, including egg cover (Welsh et al. 1993, Bury and Corn 1988).

The southern torrent salamander is a California species of special concern, and has been petitioned for federal listing under the ESA as threatened in California. In a report to the California Fish and Game Commission, CDFG has recommended that state listing is not warranted at this time (Brode 1995). The range of this species includes portions of northern California and coincides particularly with the extent of coastal forests in the northwestern part of the state (Anderson 1968). The specific habitat of southern torrent salamanders includes cold mountain streams, springs, seeps, waterfalls, and moss covered rock rubble with flowing water in humid coastal coniferous forests (Anderson 1968, Bury and Corn 1988, Welsh 1990, Zeiner et al. 1990a). An average 60m riparian buffer width was suggested to maintain suitable air and soil temperatures, and relative humidity regimes (Welsh et al. 1993, Ledwith 1996). These salamanders seem to inhabit the splash zone, and are rarely found more than one meter from water (Anderson 1968, Nussbaum and Tait 1977). As with tailed frogs, southern torrent salamanders have been linked to late seral stage forests (Welsh 1990), but observations of this species in young growth stands indicates that habitat components such as substrate, canopy closure, woody debris, and ambient temperatures may be more important than forest age (Diller and Wallace 1996, Welsh and Lind 1996, and Bury and Corn 1989). Recent research indicates that watercourse gradient and substrate type are significant habitat variables (Diller and Wallace 1996). Eggs, laid communally, hatch in 210 to 290 days, and larvae feed for 70-85 additional days while the remaining large stores of yolk are absorbed. Sexual maturity is reached 1 to 1.5 years after metamorphosis, at 4.5 to 5 years (Zeiner et al. 1990a).

The northwestern pond turtle is a California species of special concern, and a California Fully Protected Species. In California, this species ranges from the Oregon border south to Kern County (Bury 1962). The specific habitat of this species includes a variety of permanent and ephemeral aquatic habitats such as ponds, lakes, rivers, marshes, sloughs, and drainage ditches (Zeiner et al. 1990a, Bury 1962, Holland 1994, and Nussbaum et al. 1983). Aquatic habitat has been described by Bury (1972) and Reese (1996) as water < 32°C and > 0.5m deep in near shore low or no velocity stream or river reaches. Large woody debris is important as basking sites and escape habitat (Bury 1972, Reese 1996, Holland 1994). Pond turtles may use terrestrial habitats for activities including overwintering, aseasonal use, and overland dispersal (Holland 1994). At the minimum, pond turtles use terrestrial habitats for nesting (Rathburn et al. 1992, Reese 1996, Holland 1994). Females lay eggs from March to August. Female pond turtles typically excavate nest burrows in dry, compact soils high in clay or silt content. Distance of nests from aquatic habitat in one study ranged from three meters to over 402 meters (Holland 1994), and in another study averaged 50m (Reese 1996). The nesting areas tend to be on south or west facing slopes, vegetated by short grasses or forbs (Holland 1994). Three to eleven eggs are laid from March to August depending on local conditions with an incubation period ranging from 73 to 80 days. Hatchlings remain in the nest over the winter, and emerge in the spring.

Coho salmon (*Onchorhynchus kisutch*), is a member of the salmonid group of fish which are known to be highly sensitive to herbicide intoxication. The species occurs naturally in the Pacific Ocean and its tributary drainage and on the west coast of North America occurs in freshwater streams from Monterey Bay California to Point Hope Alaska (Scott and Crossman, 1973). Coho salmon typically migrate from the Pacific late in the season as fall rains increase river flow. Spawning take place in swifter waters of shallow, gravely areas of rivers during November to January. Eggs are deposited in shallow excavations in small gravel made by the female and are covered by displacement of gravel from the upstream edge of the nest. Soon after spawning is completed the adults die. Hatching takes place in early spring 35-50 days after eggs are laid. Alevins remain in the gravel for 2-3 weeks at least until the yolk is adsorbed and then emerge as free-swimming, actively feeding fry from March to late July. Although some fry may migrate almost immediately to the ocean, most remain for at least 1 year in freshwater tributaries, usually residing in gravel areas near the stream bank. In late summer or fall they move into deeper pools. Young coho in fresh water feed mainly on aquatic macroinvertebrates including Tricoperta, Plecoptera and Coleoptera.

Two neotropical migrant birds, the yellow warbler (*D. p. brewsteri* and *D. p. sonorana*) and the yelllow-breasted chat (*Ictera virens*), are considered species of special concern. Breeding populations of both species are known to occur within the AOC. The yellow warbler occurs throughout the state, excluding southeastern desert areas, and the higher mountains (Small 1974) and the breeding distribution extends across much of North America. Breeding Bird Survey results indicate that the species is decreasing in much of the Pacific Northwest and northern and central

California (Sauer et al 1997). Populations in the west have increased where the reduction of grazing and cessation of use of herbicides on willows has led to the regrowth of riparian vegetation. Breeding habitat for the yellow warbler consists of alder, cottonwood, and willow stands in riparian thickets (Harris 1991). Its feeding behaviour involves gleaning of insects and occasionally berries from vegetation. Females builds a cup-shaped nest in riparian trees generally from 1 to 60 feet off the ground (Ehrlich et al. 1988). Breeding occurs from mid-April into early August with peak activity in June. Typically, 4 to 5 eggs are laid and incubated by the female for 11 days. Both parents tend altricial young until fledging at 9 to 12 days, with the young breed the following year. Declines in the populations of the yellow-breasted chat have been observed in the southern portion of the state and are generally attributed to loss of riparian woodland, and cowbird nest parasitism (Remsen 1978). Breeding Bird Survey results indicate that the species is decreasing in the eastern US, but increasing in much of the west, including northern California (Sauer et al 1997). In California, yellow-breasted chats are found throughout the state in suitable habitat up to approximately 6,500' above sea level (Small 1974, Zeiner et al. 1990b). The specific habitat of this species is dense thickets of willow or other brushy tangles of riparian woodlands (Zeiner et al.1990b, Small 1974). It feeds by gleaning insects and occasionally berries from vegetation. Females build a cup-shaped nest in riparian shrubs or trees generally from 1 to 8 feet off the ground (Ehrlich et al. 1988). Breeding occurs from early May into early August, with peak activity in June. Typically, 13 to 6 eggs are laid and are Incubated for 11 to 15 days. Fledglings occupy the nest from 8 to 11 days thereafter and the altricial young are tended by both parents until fledging.

5.0 Herbicides and Use Patterns on PALCO Lands

5.1 General Description of Herbicides Used on PALCO Lands

The need, rationale and economic benefits associated with controlling competing vegetation in forest regeneration has been well established (Stewart *et al.* 1984; OFIA/OLMA 1990; Maclean and Morgan 1983; Perala 1982; Benzie 1977; Deloitte and Touche, 1992, Bartlett & Associates, 1989). A variety of different vegetation management techniques (e.g. mechanical disturbance, prescribed burning, herbicide application, silvicultural approaches, manual and biological methods) are available to practising foresters (Walstad *et al.* 1987). However all techniques have associated advantages and disadvantages and all carry some element of environmental risk. In addition to environmental considerations, choice of the most appropriate technique for a given situation requires consideration of numerous factors including site condition, time of year, silvicultural objectives, plant biology, environmental concerns, cost, efficacy and logistics. Owing to their high degree of efficacy, reliability and cost effectiveness, the use of synthetic herbicides continues to be a fundamental component of forest vegetation management practice both internationally and within the AOC. For example, in Canadian forestry, synthetic herbicides are applied to approximately 200,000 ha annually representing treatment of approximately 50% of the artificially regenerated land base with 76% of the treatments involve aerial application for conifer release (Campbell, 1990).

A list of herbicides currently used or expected to be used within the AOC is provided in Table 2.

Table 2. Identification and chemical content of herbicides used within the AOC

Common Name	Trade Name	Herbicide Class	Manufacturer	Active[1] Ingredient	Formulated Ingredient	
Atrazine	AATREX	s-triazine	Novartis	atrazine (480 g/L)	atrazine (480 g/L)	
Glyphosate	ROUNDUP	organophosphate	Monsanto	glyphosate (356 g/L)	IPA salt (480 g/L)	
	ACCORD	organophosphate	Monsanto	glyphosate (356 g/L)	IPA salt (480 g/L)	
Triclopyr	GARLON 4	pyridine	Dow	AgroSciences (480 g/L)	triclopyr (667g/L)	BE
	GARLON 3A	pyridine	Dow AgroSciences	triclopyr (360 g/L)	TEA salt (522 g/L)	
Hexazinone	PRONONE 10G	s-triazine	DuPont	hexazinone (100g/kg)	hexazinone (100 g/kg)	
Sulfometuron methyl	OUST	sulfonylureas	DuPont	sulfometuron (750 g/kg)	sulfometuron (750 g/kg)	
2,4-D	ESTERON 99 LV A1	phenoxy acid		2,4-D	PGBEE IOE DMA	
Imazapyr	ARSENAL	imidazolinone	American Cyanamid	imazapyr (250 g/L)	IPA salt (307 g/L)	

1. Guaranteed active ingredient, as listed on the product label

Since variable reference to herbicides by either common chemical name, trade name, active or formulated ingredient is often troublesome and confusing, herbicides will be referenced hereafter by their common names, specifying formulation type or trade names only where necessary. Distinction between active and formulated ingredients is perhaps the area of greatest confusion in herbicide nomenclature. For clarity, the term active ingredient (a.i), specifies the form of the compound which translocates and binds at the site of action to elicit a phytotoxic effect. Different formulations of the active ingredient are often used in commercial products to enhance water solubility, mixing, storage stability or other properties. Thus formulated ingredients may be chemically different than the active ingredients. However, from an environmental perspective, such differences are often unimportant, both forms are rapidly converted to the free acids upon mixing or release into the environment As an exception to this generality, the ester formulations of 2,4-D and triclopyr are important since ester forms exhibit substantially different uptake patterns in plants and substantially different toxicity in non-target organisms.

5.2 Use Rates, Application Methods & Timing

Often, the remote location and size of areas requiring vegetation management and regeneration dictate the use of aerial dispersal techniques as the most cost-effective option for herbicide application. However, given the relatively small size of cutover blocks in the AOC, the substantial topographical relief, and concerns associated with potential off-target displacement and drift, PALCO has adopted a policy of exclusive utilization of ground-based application techniques.

Ground-based techniques, as employed on the AOC are widely considered to be more environmentally acceptable, owing to reduced risk of off-target drift (Yates et al. 1978). Of the variety of ground application techniques available, those utilised on the AOC may be categorised as a) Pre-emergent, broadcast b) Post-emergent, targeted, foliar or c) Post-emergent, targeted, basal-bark. Typical application methods, rates and timing of herbicide applications are presented in Table 3.

Table 3. Typical Rates, Application Methods and Targets for Herbicides as Employed in the AOC.

Herbicide	Formulation	Avg. Rate (kg/ha ai.)	Method	Key Target
Glyphosate	ROUNDUP ACCORD	0.66 0.66	Backpack, Foliar With sulfometuron or atrazine	Pampas, roadside
Triclopyr	GARLON 4 GARLON 3A	2.5 2.3	Foliarwith atrazine Basal bark with diesel fuel Girdle, frill	Tanoak
Atrazine	AATREX 4L	4.4	Backpack, Soil With glyphosate	Competing vegetation Ceanothus
Sulfometuron methyl	OUST	0.17	Backpack, soil With glyphosate	Competing vegetation ceanothus

5.3 Use Patterns in Relation to Potential Environmental Contamination

While atmospheric, terrestrial and aquatic compartments of an ecosystem are highly interconnected, for the purpose of discussing environmental fate and risk, it is common and useful to consider them separately. Given the ground-based methods of herbicide application used on PALCO lands, studies which demonstrate that such application techniques result in minimal concentrations in air (Yates et al. 1978) and the general low volatility of herbicides under consideration here (Table 4) it is apparent that atmospheric contamination would be inconsequential. Therefore, this assessment will focus particularly on terrestrial and aquatic compartments of the environment. Since herbicide applications are intended to control competing and invasive exotic species of vegetation, the environmental constituent with highest herbicide concentrations will undoubtedly be the target plants themselves. However, some contamination of the underlying soils via either direct deposition or subsequent transport of residues via washoff and/or vegetative decay is inevitable. Further, given significant slopes, periods of high rainfall and resultant surface flow and sediment transport, there is a potential for contamination of aquatic compartments adjacent to spray sites via leaching, runoff and or mass transport mechanisms as described below.

6.0 Environmental Chemistry, Fate and Toxicology of Herbicides in Relation to Used on the Area of Concern

6.1 Overview of Herbicide Environmental Chemistry, Fate and Toxicology

Environmental fate is a phrase used to describe the cumulative and interactive results of transport, partitioning, and degradation processes which affect herbicides subsequent to their release into the environment. Environmental fate research and analyses is concerned with identifying probable environmental partitioning to air, soil, surface water, groundwater, sediment and biota, determining persistence (longevity) in each of these compartments, determining major transport and degradation mechanisms, concentrations and ultimately identifying ecosystem components most at risk. Critical elements of environmental fate analyses include characterisation of the input mechanisms, physico-chemical parameters of the chemical and parameters associated with the receiving environment which control major dissipative and/or degradative mechanisms. The typical outputs of environmental fate analyses include estimation of the time required for a compound to degrade or dissipate to 50% or 90% of its initial value in an environmental compartment (DT_{50} or DT_{90} value, respectively). Generally, the overall DT_{50} value as determined in field studies is the cumulative effect of several degradative or dissipative processes acting concurrently, although one process may be dominant. Laboratory studies conducted under highly controlled conditions are used to determine relative susceptibility to different mechanisms (e.g. photolysis, hydrolysis) and may provide rate constants characterizing the speed at which these reactions occur. As these rates are significantly affected by moisture, temperature, pH and a host of other variables they may vary substantially from rates occurring in field situations and are thus primarily useful in determining relativity's. It must be emphasised here that many degradative or dissipative functions are non-linear in nature and thus complete dissipation typically does not occur at times equal to twice that of the DT_{50} value.

Environmental toxicology refers to the qualitative and quantitative study of deleterious effects of any substance on any organism in the environment, clearly a very broad area of science and of public interest. The potential for a herbicide to induce a toxicological is a function of exposure and inherent toxicity of the compound in question. Therefore the fundamental concept of dose-response applies. This concept dictates that in order for a direct toxic effect to occur, an organism must be exposed and that progressive increases in dose generate a greater magnitude of response according to a symmetrical non-linear dose-response function. The classic "S" dose-response curve denotes two other fundamental concepts; (I) that beyond some minimal dose there is no measurable response and (ii) at some maximal dose essentially all organisms will show maximal response. It may be noted that some toxicological responses (e.g. allergic reactions and possibly carcinogenicity) may not necessarily follow the dose-response principal, however these are not directly pertinent to this assessment.

Toxicity is a relative property which refers to its potential to induce harmful effects on a living organism. Since herbicides are specifically designed to affect the physiological function of plants rather than animals, it is intuitively reasonable that impacts will occur first and foremost on terrestrial, riparian or aquatic plant species depending upon their relative degree of exposure. Substantial impact on any of these "primary producer" components may be expected to result in secondary effects on higher trophic levels, although the magnitude, duration and ecological significance of such effects is poorly understood. Notwithstanding this logic, historical toxicological research has focussed on animals, particularly mammals, fish and birds, which have historically been represented required in regulatory toxicity tests requirements. Serfis et al. (1986) (as cited in USDA-FS 1987) reviewed the use of pesticides in forestry, including the herbicides 2,4-D, glyphosate, atrazine and hexazinone, concluding that none of these chemicals exceeded the criteria for mammalian or avian species.

The potential for herbicides to elicit toxic effects on any component of the ecosystem depends upon:

a) the inherent toxicity of the compound as mediated by influencing environmental conditions (i.e. other stress factors such as temperature, pH, food limitation) and

b) exposure as determined by either direct inputs (largely dictated by method of application) and/or indirect inputs (controlled by the interaction of environmental variables with fundamental herbicide physicochemical properties as discussed previously) and

c) relative sensitivity of the organisms within the system

The most common descriptors of relative toxicity is the LD_{50} (i.e. the lethal dose for 50 percent of the organisms tested) which is generally expressed in milligrams of chemical per kilogram of body weight (mg/kg). Although toxicants are often compared based solely on their LD_{50} values, such an approach is overly-simplistic and often leads to erroneous conclusions. What is more important in assessing chemical safety is the threshold dose, and the slope of the dose-response curve, which shows how fast the response increases as the dose increases. Moreover focus on relative toxicity negates consideration of exposure as the other key determinant of potential effects. In environmental toxicology, the dose experienced by an organism is typically unknown. Most often the concentration of the toxicant in the pertinent environmental matrix is used as a surrogate for the dose. In a conceptual paradigm parallel of dose-response theory, we refer to the median effective concentration (EC_{50}) as the concentration eliciting a particular effect. In environmental toxicology, the focus is often on sub-lethal impairment of some physiological function (e.g. growth inhibition, reproductive failure) rather than mortality. EC_{50} values are generally expressed in milligrams of chemical per litre if the exposure matrix is a liquid (i.e. water) or milligrams of chemical per kg if the matrix is solid (i.e. soil or sediment). Clearly the dose encountered by any organism in either a laboratory toxicity test or in the environment will be directly dependent upon its behaviour and physiology (e.g. rate of ingestion, metabolic rate, size). As a result, in an experimental situation, all of these factors are normalised to remove them as a source of variation in the data. Unfortunately, in the real world, these and many other factors are highly variable leading to potentially significant differences in real-world toxicity as compared to what might be predicted on the basis of laboratory test results. To offset the potential influence of these variables, subjective safety factors in the order of 10 x to 100 x (Barnes and Den, 1954) have been routinely applied to laboratory toxicity data. For example, if laboratory testing showed that the EC50 for rainbow trout exposed to a pesticide is 2 mg/L and regulators wished to err on the side of safety, they established a legal water quality criterion of 2/100 or 0.02 mg/L. Currently, regulators are moving toward much more sophisticated approaches which focus on lower concentration-response endpoints (eg. EC_{10}) which more closely approximate the threshold at which toxic effects differ significantly from random effects occurring in the control population. This critical toxicity threshold value is variously estimated and may be referred to as the maximum acceptable toxicant concentration (MATC) - the hypothetical toxic threshold concentration lying in a range bounded at the lower end by the highest tested concentration having no-observed effect (NOEC) and at the higher end by the lowest test concentration having a significant toxic effect (LOEC) as determined in chronic toxicity tests (full or partial life-cycle exposures) (Rand and Petrocelli, 1985). Currently, probabilistic which compare the critical toxicity threshold endpoint, together with confidence limits about the endpoint with estimates of expected environmental concentrations and their associated confidence limits, are considered state of the art in risk analysis. Unfortunately in this case (as with many others) there is neither sufficient toxicity threshold data nor sufficient monitoring data to allow for valid probabilistic risk assessment.

6.2 Environmental Inputs, Degradation and Dissipation Mechanisms

A comprehensive review on the environmental fate of herbicides has recently been published (Grover and Cessna 1991), including an excellent chapter on dissipation and transformation processes controlling the fate and behaviour of herbicides in water and sediments (Muir 1991). Several studies demonstrate that losses to the atmosphere and atmospheric transport (drift) resulting from ground-based herbicide applications are exceedingly low. For example, under agricultural scenarios, aerial application of glyphosate by helicopter resulted in deposits of approximately 5.3% at 20 m downwind, whereas ground-rig application to the same site resulted in only 0.39% deposit (Yates *et al.* 1978). Therefore for the purposes of this assessment, where herbicides are applied via ground-based methods only, consideration of input and fate mechanisms will be restricted to direct deposition to vegetation or soils within the treated area of the terrestrial compartment with potential indirect input or movement into non-target riparian buffer zones and associated flowing aquatic systems which characterise these sites.

Degradation and dissipation mechanisms proceed according to fundamental rules of chemistry and physics. Therefore with sufficient knowledge of the basic physico-chemical characteristics of the herbicide in question and key mechanistic rates active on the molecule in the environment, overall environmental fate may be predicted with reasonable accuracy (Mackay and Stiver, 1991). The most important physico-chemical characteristics governing the environmental fate of herbicides are:

a) *Water Solubility* (WS)- the amount of chemical that can be maximally dissolved in a specific volume of water (units mg/L) provides considerable insight into the fate and transport of a chemical in the environment. In recent years, there has been a clear trend away from highly chlorinated structures to more water soluble compounds. Herbicides with high water solubility are more biodegradable, and more susceptible to rainwash from treated foliage, leaching, or movement with surface water. They tend to be non-bioaccumulating, less susceptible to volatilization, and less likely to bind to soils or sediments.

b) *Octanol/Water Partition Coefficient (K_{OW})* - the ratio of chemical concentration in octanol versus that in water as observed in standardized laboratory tests using defined solvent volumes. Often expressed as a logarithm, as a ratio the parameter is dimension less. K_{ow} has been proven to be highly correlated to bioconcentration in aquatic organisms as well as soil sorption and thus may be used as a predictive tool in estimating these two potentialities.

c) *Dissociation Constant (pKa)* - the acid dissociation constant is the pH at which a compound is in equilibrium between its dissociated and molecular states. The degree of dissociation is particularly important in the "ion trapping" phenomenon via which weak acid herbicides (2,4-D, triclopyr, imidazolinones, sulfonylureas) are phloem loaded and also has significant effects on soil sorption, bioconcentration, evaporation and photolysis. The behaviour of glyphosate, a zwitterionic (i.e. having both positive and negative charges) molecule with four different pKa values, is almost entirely directed by pH.

d) *Partition Coefficient to Organic Carbon (K_{OC})* - the partition coefficient to organic carbon is the ratio of chemical concentration sorbed to organic carbon when shaken in a solution of water and is dimension less. Closely related to K_{ow}, the K_{oc} value provides a more definitive assessment of potential for herbicides to bind to organic matter in soils and other terrestrial matrices, sediments and biota.

e) *Soil Partition Coefficient (K_d)* - The soil adsorption coefficient is determined experimentally by mixing pesticide with a slurry of soil and water, shaking for long enough to allow the pesticide to partition between the water and soil (equilibrium) and then centrifuging to separate the water and soil. The concentration of pesticide in the water is then determined and the ratio of concentrations in the two phases calculated to give the adsorption coefficient, K_d. The degree of sorption to any soil is a function of both the soil type (primarily % organic matter, % clay and cation exchange capacity) and the chemical nature of the herbicide.

f) *Bioconcentration Factor (BCF)* - The bioconcentration factor is the ratio of the concentration of a chemical within an organism to that of its surrounding environment at equilibrium. Certain chemicals, due to their hydrophobic nature have a tendency to partition through biological membranes and concentrate within tissues. Low water solubility and high K_{ow} values are indicative of substances with this tendency and thus BCF and Kow are highly positively correlated while BCF and WS are strongly inversely correlated (Veith, 1979).

g) *Vapour Pressure (VP)* - Vapour pressure is the pressure exerted by the vapour of a substance when at equilibrium with it s liquid or solid phase at some constant temperature. Compounds with vapour pressures less than 10^{-6} are essentially non-volatile and tend to remain adsorbed to surfaces or dissolved in matrix.

A review of the physico-chemical properties for many herbicides appears in Bailey and White (1965) and a comparison of some important characteristics for herbicides under consideration here is provided in Table 4, from which a number of important generalities can be drawn:
a) the majority of the herbicides considered in this assessment are moderately to highly soluble in water, only ester forms of 2,4-D and triclopyr are characterized by low water solubility (3-6 orders of magnitude less soluble than acid and salt forms)
b) the majority of herbicides considered in this assessment are characterized by low Kow values, again with the exception being ester forms of 2,4-D and triclopyr which have high Kow values and a proportionally greater tendency to partition to lipids or organic matter
c) most herbicides have pKa values less than 3.5 and thus at typical environmental pH (5-8) these compounds are predominantly anionic or negatively charged and will tend to sorb to positively charged surfaces of organic matter and clay or bind with cationic metal ions within matrices
d) soil adsorption coefficients vary markedly among herbicides and across soil types, however soils with finer texture, higher clay and organic matter content tend to exhibit maximal adsorption
e) with the exception of the esters of 2,4-D and triclopyr, herbicides considered in this assessment show essentially no propensity to accumulate in aquatic organisms.
f) for most herbicides considered in this assessment, vapour pressures are well below threshold values (10^{-6}) of volatility and thus volatilization would be expected to be minimal under all but extreme conditions of temperature, windflow and turbulent mixing. Exceptions are the esters of 2,4-D and triclopyr where volatilization losses may be important under certain conditions.

The ultimate fate of a herbicide released into the environment is dictated by the interactive effects of dissipation and degradation mechanisms. To distinguish, degradation involves a change in molecular structure typically leading to simpler, more water soluble and more labile products, whereas dissipation (which may also significantly reduce the concentration of a parent compound in a particular environmental compartment) results from dispersal, sorption or transport without a change in molecular structure of the herbicide. A summary of primary degradation mechanisms and principal metabolites for herbicides under consideration is provided in Table 5.

Table 4. Fundamental physicochemical properties of herbicides used in Canadian forest vegetation management.

Herbicide	MW	WS (mg/L)	K_{OC}	Log K_{OW}		BCF	pKa VP (mm Hg)
Atrazine	215.7	33	25-155	2.75	<12[a]	1.7	2.8×10^{-7}
Glyphosate	169.1	12,000	3×10^{4}	-4.6	3	0.8, 2,	0.04mPa
IPA	228.2					6,10	
Triclopyr							
Acid	256.5	440	27	-0.45	0.5	2.68	0.168mP
ester	356.6	0.2	1200	1.2×10^{4}	400		
amine	371.1						
Sulfometuron methyl	364.4	300	171	-0.51		5.2	73fPa
Hexazinone	252.3	33,000	0.2-1	1.2	5-7	1.09-1.23	8.5mPa
2,4-D							
acid[1]	221	0.9	20	330-617	13	2.7	6×10^{-5}
ester		0.9					6×10^{-6}
amine	3×10^{6}						
Imazapyr							
acid	261.3	11,272	1.7-4.1	0.11	low	1.9,10.5	1 x 10-7
IPA	320.4						

Sources: Howard (1991), Worthing and Hance (1991)
Where: MW = molecular weight, WS = water solubility in mg/L, Koc = organic carbon partition coefficient, Kow = octanol:water partition coefficient, BCF = Bioconcentration factor, pKa = acid dissociation constant and VP = vapour pressure in mm
a general values for most species as summarized in Solomon et al. 1996, substantially higher values for mayfly nymphs (480) and soil fungi and bacteria (87-132) were reported.

Table 5. Primary mechanisms of degradation for herbicides used within the AOC

Herbicide Ingredient	Primary Mechanism of Degradation: Plants	Soils	Water	Major Metabolite
Atrazine	hydroxylation	microbial	microbial	2-hydroxy atrazine
Glyphosate	conjugation	microbial	microbial	aminomethylphosphonic acid (AMPA)
Triclopyr	hydrolysis & conjugation	microbial	photolysis	trichloropyridinol (TCP)
Hexazinone	hydroxylation demethylation	microbial	photolysis	hydroxycyclohexyl & N-demethyl derivatives
Sulfometuron methyl	hydrolysis	hydrolysis	hydrolysis	methyl-2(amino-sulfonyl) benzoate
2,4-D	hydrolysis & conjugation	microbial	microbial	dichlorophenol
Imazapyr	conjugation	hydrolysis	photolysis	2,3-dicarboxypyridine

a) Biodegradation

Herbicides released into the environment, are subjected to both biologically mediated (biotic) and non-biological (abiotic) degradation processes. The importance of microbial degradation reactions is not surprising given the diversity and uniqueness of their metabolic activity, as well as the variety of environments in which they thrive. The latter also attests to their adaptability and tolerance levels, and indeed it is this characteristic that allows microbes to adapt to pollutants and use them as carbon, nitrogen or other elemental sources. Microorganisms are key agents in the degradation of a vast array of organic pollutants in both terrestrial and aquatic compartments. Biodegradation reactions are typically characterized by curvilinear kinetics (i.e. changing rate constants with time) which may result from a number of different factors (e.g. microbial adaptation and rate controlling desorption of pollutant molecules from soils or sediments). Several mechanisms for biotic degradation and transformation may be differentiated - metabolism, cometabolism, polymerization and conjugation. Metabolism is a term generally used to denote the case in which the pollutant can serve as a substrate for microbial growth. Typically, metabolism results in complete mineralization to CO_2 and other organic components from which the microbe can attain required energy. Cometabolism is the more prevalent form of microbial degradation in the environment and involves the conversion of a compound without the microbe deriving any nutrition or energy from it. In essence, cometabolism is fortuitous, resulting from enzymes which lack substrate specificity. Cometabolism typically does not result in complete mineralization. An interesting phenomenon associated with biological degradation by microbes involves their rapid population turnover rates and adaptation. Under some conditions, for some compounds, pollutant concentrations may induce a selective pressure on microbial populations to favour species or induce enzymes utilizing the pollutant as a substrate. In these situations, pollutant degradation typically follows a lag phase (while populations adapt or specific enzyme levels increase) followed by a log phase (where pollutant degradation occurs at much faster rates owing to the adapted population). Biodegradation of phenoxy herbicides in soils is considered the classical example of this phenomenon. Biological degradation is a critical determinant of overall herbicide fate and persistence for any compound with an inherent tendency for biologically mediated reaction and for any environment in which microbial (or other biodegrading organisms) have high populations. As such compounds that are highly halogenated are less susceptible to biodegradation and this process is relatively less important in dry soils, cold conditions, or in the atmosphere where microbial or other biological populations and activities are low. Biodegradation may occur through a host of different reaction mechanisms involving different enzymes that mediate specific reactions such as reduction, oxidation, hydrolysis, decarboxylation, demethylation, hydroxylation, epoxidation, reductive dechlorination.

b) Photolytic degradation - sunlight is one of the primary forces affecting the loss of herbicides from atmospheric, exposed terrestrial or aquatic surfaces. Considered as the process in which ultraviolet (UV) or visible light causes transformation of compounds, sunlight together with other environmental reactants can chemically degrade pollutants with rates and products varying considerably depending upon the compartment in question. While photolysis may be the key degradative pathway for a compound in one compartment, it may not occur at all in another compartment (eg. sorbed compounds buried at depth in soil are not susceptible, compounds absorbing at certain wavelengths may not be photolytically degraded in water which attenuates that wavelength). Thus assessing the importance of photochemical degradation processes, requires not only an understanding of the basic photochemistry of the xenobiotic but also and understanding of the environmental chemistry of the compartment in which it resides. Again the relative importance of photolysis must be considered in relation to all other degradative mechanisms. Of all environmental compartments, photolysis in aquatic systems has received the most attention. Leaders in this field include Crosby and Wong e.g. [1977], Zepp [e.g.1978] Leifer and Zafiriou [e.g. 1984]. Photolysis may occur by either direct or indirect mechanisms. Direct photolysis requires absorption of light by the xenobiotic molecule, the excess energy breaks bonds and generates photo products. Direct photolysis depends upon the overlap between the sorption spectra of the pollutant molecule and the spectral distribution of light striking it. Radiation at wavelengths below 290 nm are efficiently absorbed by ozone in the atmosphere, hence, compounds with a sorption maxima less than 290 nm will typically not undergo direct photolysis. In estimating direct photolysis rates, one must consider that irradiance changes with latitude, season, time-of-day and degree of attenuation (i.e. by ozone, molecular scattering as in water, particulate diffusion). Fortunately, comprehensive models (e.g. SOLAR) have been devised which estimate the solar irradiance intensity for critical wavelengths at any time or position on the globe

Indirect photolysis begins with light being absorbed by some substance other than the xenobiotic in question. A chain reaction series is invoked which ultimately results in the photolysis of the xenobiotic molecule. Naturally occurring organic and inorganic species absorb radiation required for the subsequent reactions, most commonly these are humic substances, clay minerals, and transition metals. Oxidation is the predominant indirect photolytic process, with oxidants activated primarily by two processes; energy transfer to oxygen producing singlet oxygen ($^1O_2$)and radical production, that in turn results in the formation of radical as well as non radical active oxidants The products of direct or indirect photolysis mechanisms may be the same, but the kinetics are often different.

Photolysis on soil surfaces is poorly understood. Unlike solutions, surfaces may be highly heterogeneous and may result in incident light which is difficult to characterize. Of course soil and waxy leaf surfaces attenuate light strongly, thus limiting the degree to which photolysis can occur to only those molecules at or near the surface proper. The effect of soil properties including moisture content, pH and Eh are critical determinants of photolysis as well as other abiotic degradation processes in soils.

C) Hydrolytic Reactions - in general the term hydrolysis refers to the cleavage of a bond of the xenobiotic and the formation of a new bond with the oxygen atom of water. The term is often loosely (and erroneously) used to describe all reactions in which water serves as a solvent. Many of these (eg. elimination, decarboxylation, isomerization) do not incorporate water in to the transformation product and are therefore not hydrolytic reactions. Hydrolysis reactions are one of the most important abiotic degradation processes for compounds in aquatic environments and are generally categorized depending upon the type of pH catalysis. In acid-catalysed hydrolysis, an acid (usually a proton H+) enhances the process but is not consumed in the reaction. Since the rate of the reaction is dependent upon the proton concentration, acid-catalysed reactions increase as the pH decreases. Similarly, base-catalysed reactions are enhanced by hydroxyl (OH-) ion concentrations and exhibit increased reaction rates at higher pH levels. Some hydrolytic reactions are independent of pH. Given sufficiently drastic conditions, many xenobiotics will undergo hydrolysis, however the contribution of hydrolysis to overall degradation depends upon the relative rate of reaction with respect to other competing reactions (e.g. photolysis, biodegradation etc.) under the pertinent environmental conditions. Functional groups that are highly susceptible to hydrolysis include carboxylic acid esters, organophosphates and their esters, amides, anilides, carbamates, organohalides, triazines, oximes and nitriles. Organohalides may lose their halogen substituents by a myriad of pathways, some of which are hydrolytic and with varying pH dependencies [Jeffers et al. 1989]. In natural waters, hydrolysis may occur in the dissolved phase or at the water-sediment interface and at generally equal rates.

d) Redox Reactions - Although many oxidation reactions that occur, particularly in surface waters, are believed to be photolytically or biologically mediated, various pathways for direct oxidation of pollutants are also known. A variety of oxidizing agents occur in the environment including monooxygenases, dioxygenases, peroxidases and lactones. Of the various reducing agents available Fe and Mn species are most important, although bioorganics such as flavins, porphyrins and extracellular enzymes may also be responsible for reductive reactions, particularly in natural waters. Next to hydrolysis, redox reactions art thought to be one of the most dominant processes in aquatic environments occurring not only in the dissolved phase but also within sediments. Surface effects as well as restricted diffusion of O2 through the sediment may dramatically affect the rates of the redox transformations that occur in sediments.

e) *Polymerization and Conjugation* - Polymerization is a transformation process in which coupling reactions take place. In the case of herbicides, conjugation reactions, in which residues are bound to substrates such as sugars (glucose in plant systems and glycogen in animals) are most important. Other substrates such as glutathione, ribose, glucorinic acid and sulphate and phosphate ions may also be involved. Conjugation reactions are a key mechanism for herbicide detoxification in both plant and animal systems (Stephenson and Solomon, 1991).

Thus both biotic and abiotic degradation of herbicide residues may occur simultaneously in all components of both the terrestrial and aquatic environments of the AOC. It is difficult if not impossible to generalize on the relative importance of the two types of transformation processes. However, biotic degradation pathways tend to dominate when moisture, microbial and organic matter content are high. Dissipation mechanisms which may be active on herbicides applied within the AOC are dependent upon where initial deposits occur. In foliar or basal bark applications, a substantial amount of the applied chemical must be taken up and translocated to the site of action within the plant in order for silvicultural objectives to be met. For herbicide impinging on the leaf surface, the plant cuticle acts as a primary barrier to herbicide penetration. Studies typically show that for most herbicides less than 50% of the applied amount applied penetrates the cuticle and enters the plant leaf cells. Formulation characertistics are key determinants of cuticular penetration, with ester formulations (e.g. 2,4-D iso-octyl ester and triclopyr butoxyethyl ester) penetrating more readily than amine formulations. Residues remaining on the plant leaf surface are subject to a variety of dissipation and degradation mechanisms including rainwash, transport with leaf-fall, volatilzation, photolytic, microbial and hydrolytic degradation as described below. In a similar fashion basal bark treatments result in the majority of the herbicide depositing on the bark surface. Since water-soluble chemicals or aqueous spray solutions penetrate the bark poorly, oil-soluble ester or amine formulations mixed with petroleum-based carriers are used to enhance uptake which occurs principally through lenticels or pores in the bark or through natural breaks in the bark such as growth cracks. Quantitative estimates of uptake rates following herbicide deposition on bark are not available. However, the high degree of phytotoxicity typically resulting from such treatments suggests that substantial uptake does occur through this mechanism. As for foliar deposits, herbicide residues remaining on the bark surface would be susceptible to rainwash, volatilzation, photolytic, microbial and hydrolytic degradation, but to an arguably lesser degree given the cooler shaded conditions which may be assumed for basal portions of most woody stems.

Depending on density, leaf-surface area and degree of spray retention, vegetation within the treated site mediates against direct herbicide deposit to the soil surface. Similarly, large woody debris which remains may intercept and sorb differential amounts of applied herbicide. Subsequently, indirect deposition to soils may occur through wash-off of residues on foliage or woody debris, via leaf-fall or via plant-decay. Indirect inputs to soils via washoff may be particularly important for more water soluble, ionic compounds characterized by low K_{ow} value and where rain or dew fall occurs shortly (<24-36 hrs) after chemical applications are made. Once in the soil compartment, herbicide fate is controlled principally by its inherent physico-chemical properties and by the adsorption-desorption process. As described previously in section , herbicide sorption in the soil compartment is largely a function of organic matter content and cation-exchange capacity. However, the adsorption-desorption phenomenon may be influenced by a variety of other factors which in turn indirectly influence overall soil fate, as shown in Table 6.

Table 6. Factors Affecting the Adsorption-Desorption of Herbicides in Soils

Factors resulting in greater desorption	Desorbed Herbicides are more likely to
Higher soil temperature.	Volatilize from the soil.
Higher pesticide solubility within related groups.	Move downwards by leaching.
Higher soil moisture in light soils.	Move laterally with runoff water.
Greater percent sand.	Be degraded by microorganisms.
Higher soil pH (7-12).	Be taken up by crops in the soil.
Factors resulting in greater adsorption	**Adsorbed pesticides are more likely to:**
Higher clay content.	Move with eroded soil.
Higher organic matter content.	Be degraded chemically.
Greater polarity of the pesticide molecule.	Be taken up by soil animals such as earthworms (if lipophilic).
Ionic sites on the pesticide molecule.	

In forest herbicide applications, indirect herbicide inputs into aquatic ecosystems may result from mobilization of poorly-sorbed residues on treated foliage (rainwash) or in ephemeral stream channels, and as surface runoff (dissolved or bound to particulate) or through downward percolation (leaching). Such non-point source inputs are similar to those of agricultural systems (Wauchope *et al.*1978; Haith 1985; Grover and Cessna 1991). Ultimately, leaching or offsite lateral movement may result in contamination of groundwater and/or surface waters

In aqueous environments there are three key processes of mass transport - advection, dispersion and sorbed phase transport. Advection refers to transport of dissolved substances by the water current in any of three directions (longitudinal, lateral, and vertical). Dispersion refers to mixing throughout the water column which can result from simple or turbulent diffusion and which may occur in all three directions as noted above. In natural waters and at environmentally relevant concentrations, herbicide residues may exist in either dissolved phase or sorbed to fine suspended material, depending upon their physicochemical characertistics. Sorbed phase transport refers specifically to transport of herbicides associated with such material which becomes entrained in the current and moves at the same velocity as the water itself. Advection, turbulent diffusion and sorbed phase transport mechanisms are all active in flowing aquatic systems resulting in rapid dissipation of chemical concentrations and substantially reduced exposure periods for aquatic biota. In contrast, simple diffusion predominates in groundwater as well as slow or non-flowing surface water bodies. As a general rule, shallow, slow-flowing or non-flowing water bodies, where dilution and dispersion mechanisms are minimized are considered a worst case scenario. Where these conditions are couple with low sunlight irradiance (reduced photolysis), cooler temperatures (potentially slower microbial and hydrolytic degradation rates) and minimal organic matter content of sediments (reduced adsorption) exposures may be further exacerbated.

6.3 Environmental Chemistry, Fate and Toxicology of Atrazine

Atrazine was the first member of the symmetrical triazine class of herbicides introduced by the Ciba-Geigy corporation and was registered in the United States in 1959. Since that time it has been extensively used as a selective herbicide in agriculture, particularly in corn and sorghum production. Although a number of formulations and trade names are available, the liquid formulation marketed as AATREX 4L is most commonly employed in forestry and within the AOC. Atrazine is used for non-selective weed control in site preparation and to a lesser extent for conifer release, management of wildlife habitat, noxious weed control and on rights-of-way (Gross, 1983). A detailed review of atrazine in relation to forest uses is available (USDA-FS, 1984), which documents typical application rates ranging from 0.6 to 5.5 kg/ha a.i. Application rates as employed within the AOC (4.4 kg/ha a.i.) are at the upper end of this range. Based on its physico-chemical characteristics (Table 4), atrazine may be characterised as a herbicide with moderate water solubility, low vapour pressure, and low potential to sorb to soils or partition and bioaccumulate in animal tissues.

In plants, atrazine is taken up principally through the root system (Burken and Schnoor 1997) but limited foliar uptake is also known to occur. Its phytotoxic mode of action is dependent upon specific binding to a 47 kilo-dalton protein of the plant chloroplast through which it inhibits the electron transport chain of photosynthesis. Within the plant, atrazine moves easily with the transpiration stream. As such, factors such as higher temperatures and lower relative humidity which increase transpiration rates also accelerate atrazine translocation. Plants have widely differing capabilities to degrade atrazine and this differential is the basis of atrazine selectivity. Degradation mechanisms active on atrazine molecules *in planta* include hydrolysis, conjugation, dealkylation and oxidation, however hydroxyatrazine is typically the dominant plant metabolite (Esser et al. 1975). Although vapor pressure is low, substantial volatilization losses from treated foliage have been reported (Burt, 1974). Residue levels for plants following forestry applications are not available for atrazine. However, based on ratios of plant residues to application rates (100 mg/kg per kg/ha applies) as estimated by Newton and Dost (1981) or by Thompson et al (1994) (313 mg/kg per kg/ha applied) would generate estimates of plant residues in the range of 440 to 1377 mg/kg. Determining the biological significance of such residues is a difficult exercise in extrapolation. However, assuming worst case scenarios of such maximal residue levels in vegetative food materials and that these are constant (i.e. no degradation occurs) and that exposures are by oral ingestion only, residue concentrations in vegetation can be compared to concentrations in feed which induce lethality to 50% of test populations under laboratory toxicity testing protocols. In the case of atrazine the 1377 mg/kg estimate for foliar residues is substantially (~3.6 x) lower than LC_{50} values of 5,000 mg/kg reported for several bird species (bobwhite, japanese quail, ring-necked pheasant, mallard) by Hill et al. 1975, suggesting little potential for direct effects on avian species.

Following deposition to soils, atrazine is susceptible to photodegradation, hydroxylation and dealkylation reactions (Jordan et al. 1970) as well as microbial degradation. Microbial degradation of atrazine has been reviewed by several authors (Esser et al. 1975; Ghassemi et al 1981; Kaufman and Kearney 1970; Parris and Lewis 1973). Kaufman and Kearney (1970) list four species of bacteria and 21 fungi that have been shown to metabolize atrazine. Hickey et al (1994) have isolated another soil microbe with atrazine degrading capability. The three pathways for microbial degradation include dealkylation, hydrolysis and cleavage of the triazine ring (Ghassemi et al. 1981). Soil conditions which favour microbial growth (higher temperature, moisture and organic matter content) enhance atrazine degradation rates (Esser et al 1975). Notwithstanding demonstration of microbial degradation of atrazine in soils, abiotic hydrolysis and photodegradation are considered to be the principal degradation mechanisms in natural soil environments. Organic matter such as humic acid and low pH are known to catalyze abiotic hydrolysis reactions. The principal metabolite, hydroxyatrazine, has been shown to be more persistent in soils than the parent compound (Smith 1982, Jones et al. 1982).
Based largely on studies conducted on agricultural soils, atrazine is considered as moderately persistent, with DT_{50} values ranging from 90 to 365 days (USDA-FS, 1984), similar values for aerobic and anaerobic soil metabolism studies in California loam and sandy loam soils were 146 and 77 days respectively. Although atrazine is only moderately soluble in water it has a relatively low affinity for binding to either organic carbon (Koc =25 to 155) or soils (Kd 0.19 to 2.46) (Ciba-Geigy, 1994). Laboratory studies demonstrate moderate mobility in silty clay loam soils (Helling, 1970, 1971) and it is known to be susceptible to leaching (Lavy 1970). Atrazine mobility is known to be concentration dependent with higher concentrations being more mobile in silty clay loam, sandy loam and fine sand soils (Davidson 1979; Davidson et al. 1980). Although atrazine has a propensity to move in dissolved phase and is susceptible to leaching, the actual amount of soil-applied atrazine leached to substantial depths in soil (> 163 cm) is quite low and < 1% is of total applied atrazine is typically recovered in tile drainage water from agricultural sites (Von Stryk and Bolton, 1977, Muir and Baker, 1978). A number of studies have examined the leaching behaviour and runoff potential for atrazine but few are directly pertinent to use patterns in forestry. Estimates of runoff losses from studies conducted in agricultural scenarios range from 0 to 15% of the applied material (USDA-FS, 1984), with most studies documenting losses of < 5% (Foy and Hiranpradit 1977; Ritter et al. 1977; Gaynor and Volk 1981; Hall et al. 1972; Rohde et al. 1981). Soil adsorption of atrazine is maximal in soils with higher clay

and organic matter content (Williams, 1970; Hayes 1970; Adams and Pritchard 1977). Two studies (Dao 1979; Scott and Phillips 1972) have reported that atrazine adsorption increases with decreased soil moisture. Forest soils within the AOC are higher both in high organic matter and clay content than agricultural soils (OM% = 0.8 to 4.8%) for which partition coefficients of (Kd = 0.19 to 2.46, Koc=25.3 to 155.0) have been estimated (Ciba-Geigy 1994). Studies reported in the latter reference clearly demonstrate greater adsorption of both atrazine and its principal metabolite, hydroxyatrazine, in clay and fine texture soils as compared to coarse textured sands. Thus, stronger binding as well as lower leaching and runoff potential may be expected as compared to agricultural soils . Further, while slopes of application sites within the AOC are undoubtedly greater than those of agricultural sites in the mid-western USA where most studies have been conducted, enhanced potential for runoff with greater surface flow may be effectively offset by resorption either by vegetation, woody debris or charcoal from prescribed burns within the site or in the vegetated buffer zones bordering all Class I and II streams. Baker et al (1982) reported that retention of corn residues on agricultural fields resulted in 17 to 59% reductions in loss of atrazine under moderate rainfall conditions, with maximal residual crop cover resulting in minimum off-site movement of atrazine. Similarly in a review of 13 trials by Fawcett et al. (1994), atrazine losses from fields under conservation tillage were < 44% of those under conventional tillage. These studies suggest that in cases where atrazine is applied to forest sites with substantial vegetative cover, large woody debris, and bounded by vegetative buffers of 100 to 170 feet, inputs to adjacent Class II and I streams should be minimal.

In aqueous environments, degradation of atrazine occurs slowly in the absence of sediments, for example, Weidner reported DT_{50} values for atrazine of > 15 months in ground water at 10 °C. In surface waters, abiotic degradation via sediment and acid catalyzed hydrolysis is considered the major degradation pathway (Kneusli et al. 1969, Ciba-Geigy Corporation1994, Khan, 1978, Li and Felbeck, 1972, Armstrong et al 1967). Although marine fungi capable of degrading atrazine are known (Schocken and Speedie, 1984; Schocken et al, 1982) and bacterial enrichment of aqueous solutions has been shown to enhance atrazine degradation (Geller et al 1980), biodegradation in most natural environments occurs slowly and is of minor importance. Owing to its low vapor pressure, volatilization of atrazine from water surfaces is predicted to be an unimportant loss process (Muir, 1991). While photolysis of atrazine does not occur in water at wavelengths greater than 300 nm, rapid photodegradation has been reported at 290 nm and is further accelerated in the presence of photosensitizers (Pape and Zabik, 1970) Both hydrolysis and photodegradation lead to formation of the principal metabolite hydroxyatrazine (Khan, 1978; Wolfe et al., 1976; Burkhard and Guta 1989) which has a greater affinity for sediments than the parent compound (Lay et al 1984). Jones et al (1982) compared persistence of atrazine in estuarine water and sediments under both aerobic and low oxygen conditions in laboratory studies. The reported a relatively rapid loss of atrazine in water (DT_{50} 3 to 12 days) and sediments (15 to 20 days).

The occurrence, fate and potential effects associated with atrazine in surface waters have recently been reviewed in a probabilistic risk assessment pertaining to agricultural drainages in the mid-western United States where atrazine has been repeatedly and extensively used for approximately the last 40 years (Solomon et al., 1996). As noted by these authors, pesticide exposure in river and stream ecosystems can be influence greatly by stream order and watershed size. Smaller watercourses tend to be characterized by higher concentrations, owing to minimum water volumes and thus dilution, but for relatively short periods as a result of minimum hydrologic response times. In contrast, higher order streams and rivers draining larger watersheds may exhibit lower magnitudes of concentration (dilution effect of high water volumes) but for longer periods (greater hydrologic retention). The study of Solomon et al (1996), demonstrated that under these agricultural scenarios, 90th percentile values for instantaneous concentrations of atrazine ins streams and rivers was < 0.0042 mg/L, while 4-day and 21 day values were < 0.01 mg/L. Further, they noted that atrazine concentrations in water were essentially non-correlated with suspended solids (r^2=0.06), or nutrient levels in streams. In relation to forest-use scenarios, it should be noted that historically agricultural applications of atrazine have been made to bare soils, where fields may be underlain by drainage tiles running directly into watercourses, and where fields may extend right to the edge of rivers or streams with little or no buffer.

Atrazine is moderately to highly toxic to a wide variety of plant species and nontoxic or slightly toxic to a variety of animal species (USDA-FS, 1984). Studies summarized in the latter review indicate that atrazine is nonteratogenic, has little or no effect on fertility, reproduction or development of offspring, and is non mutagenic. Dietary studies and egg injection bioassays with atrazine have been summarized and indicate a low toxicity to birds (USDA-FS, 1984). Results of 8-day feeding studies with bobwhite quail indicated LC_{50} values ranging from 700-800 mg/kg (Heath et al. 1972) while other studies have documented LC_{50} values greater than 5,000 mg/kg for bobwhite and Japanese quail, as well as ring-necked pheasants. Hoffman and Albers (1984) reported on potential embryotoxicity and teratogenicity of 42 herbicides, insecticides and petroleum contaminants to mallard eggs, found 2,4-D, glyphosate, and atrazine to be only slightly or nontoxic (i.e. LC_{50} values equivalent to 196 to 550 kg/ha). In their comprehensive risk assessment, Solomon et al. (1996) have recently summarized the substantial aquatic toxicity database for atrazine. Their results indicated that acute or chronic data was available for 85 aquatic species in freshwater and several saltwater species. Based on comparison of geometric means for acute toxicity tests, phytoplankton with mean EC50 concentrations ~ 0.1 mg/L were most sensitive, followed by aquatic vascular plants (macrophytes mean EC50 ~ 0.2 mg/L), benthos (mean EC50 mg/L ~ 1.4 mg/L). Fish and amphibians were relatively least sensitive to atrazine intoxication, representative 96 hr LC_{50} values are given in Table 7. As commonly observed in toxicity studies in fish, salmonid species tended to be most sensitive. The lowest 96 hr LC_{50} value for fish was 3.5 mg/L reported for rainbow trout (*Salmo gairdneri*) (Bathe et al. 1976). MATC, NOEC, and LOEC values reported for freshwater fish species ranged dramatically from 0.06 to > 10 mg/L. A NOEC value of >0.34 mg/L was reported for *S. gairdneri* exposed for 10 days (Davis et al. 1994) and a MATC value of 0.06 to 0.12 mg/L was reported for *S. fontinalis* exposed for 96 hrs.

Toxicity studies on atrazine have also been conducted for a variety of amphibians, including egg and early tadpole stages of both frogs and toads. These studies (Table 7) indicate LC_{50} values ranging from a low 0.41 mg/L for the bullfrog (*Rana catesbeiana*) to greater than 48 mg/L for the American toad (Birge et al 1980; 1983). With the exception of the low estimate cited for bullfrogs, LC_{50} values for amphibian eggs and tadpoles were greater than 5 mg/L and thus similar to those of the most sensitive fish species. Teratogenic effects of atrazine have been demonstrated in both fish and frogs, with spinal cord and other skeletal malformation in fish fry and in frog tadpoles that were hatched from eggs exposed to lethal concentrations (Birge et al. 1979; 1980; 1983). In channel catfish eggs exposed to

4.83 mg/L of atrazine, 69% of hatchlings showed teratogenic effects, similarly 62% of rainbow trout eggs exposed to 5.02 mg/L resulted in malformed fry. In amphibians, teratogenic effects were observed in 97% of hatched bullfrog eggs exposed to 24.4 mg/L atrazine and in 17% of eggs of American toads exposed to 48.2 mg/L. It should be noted that all of the teratogenic effects observed for atrazine in fish and amphibian species were induced by concentrations in excess or equivalent to LC_{50} values for these species. Thus, mitigation of exposures to protect against lethality should also be protective against potential teratogenic effects.

6.4 Environmental Chemistry, Fate and Toxicology of Glyphosate

Glyphosate, is an organophosphate herbicide, discovered by the Monsanto company and registered in 1974. It enjoys widespread application in agriculture, industrial, noxious-weed, home-use and forestry vegetation management scenarios (Gross, 1983). The recent introduction of transgenic crop plants (e.g. soybeans and cotton) resistant to the phytotoxic action of glyphosate, will undoubtedly broaden its use in agriculture. A detailed review of glyphosate in relation to forest uses is available (USDA-FS, 1984) which documents typical application rates ranging from 1.1 to 5.5 kg/ha a.i. in either site-preparation or conifer release treatments. Historically, a single formulation of glyphosate containing the isopropylamine (IPA) salt (in the USA under tradename ROUNDUP, in Canada as VISION) has been marketed for use in forestry. More recently, new formulations such as ACCORD which also contain the IPA salt but no surfactant have been developed and used. A detailed review of all aspects glyphosate development and use is available (Grossbard and Atkinson, 1984). Glyphosate is characterised by a high water solubility, low vapor pressure, low Kow, and low BCF value, indicating that it is unlikely to volatilize significantly from treated surfaces or water and unlikely to bioaccumulate in animal tissues. The high Koc value reported for glyphosate reflects its zwitterionic or dual electrical charge structure which induces exceedingly strong binding affinity to organic and clay constituents of soils. As a result glyphosate exhibits essentially no phytotoxicity when applied to soils, is very resistant to desorption therefrom and thus not susceptible to leaching or movement with surface water. In the environment, although photolysis has been demonstrated, glyphosate is degraded principally by microbial mechanisms, yielding the principal degradation product - aminomethylphosphonic acid (AMPA). Glyphosate is highly toxic to a wide variety of plant species but essentially nontoxic to most animal species (USDA-FS, 1984). Studies summarized in the latter review indicate that glyphosate is nonteratogenic, has little or no effect on fertility, reproduction or development of offspring, is non mutagenic and non-carcinogenic. Sullivan and Sullivan (1990) provide a compilation of studies documenting potential non-target effects of glyphosate, including 37 published studies on potential effects to small mammals. While detailed discussion of these studies is beyond the scope of this document, collectively these studies support the conclusion that physiological changes in individual animals which may result from ingestion or exposure to glyphosate do not induce changes in demographics which are apparent at the population level (Sullivan, 1994).

In plants, glyphosate is a potent inhibitor of the shikimic acid pathway of amino acid synthesis (Stasiak et al. 1991). Brecke and Duke (1980) have demonstrated that glyphosate rapidly penetrates plant cuticles (within a 4 hr time frame), but is only slowly taken up by mesophyll cells. In reviewing the uptake and translocation processes governing the behaviour of glyphosate *in planta*, Caseley and Coupland (1985) suggest that glyphosate uptake is a biphasic process with initial rapid penetration, followed by a slower phase. The duration of either step in this process is dependent on a number of factors including species, age, environmental conditions, concentration of glyphosate and concentration of the surfactant. This, and similar results from other researchers, has led to the suggestion that slow symplastic uptake is the rate limiting, and that the biphasic pattern may be a key element in the excellent systemic activity observed for this compound. The addition of surfactants in glyphosate formulations facilitate uptake through the plant foliage, however, the uptake process is slow and significant residues of glyphosate may remain on the leaf surface being susceptible to washoff with rain. Lund-Hoe (1976) reported on the uptake, distribution and metabolism of this herbicide in spruce and later in two brush species - ash and birch (Lund-Hoie, 1979). In the latter study, authors reported that only 20% of the glyphosate initially applied was absorbed in ash and birch foliage and that decomposition of the absorbed chemical was slow (35% in 2 months). Studies by Leung and Webster (1993) indicate that in leaves of trembling aspen, 62% of initial foliar deposits were recoverable in leaf washes at 36 hrs post-treatment and significant losses (55%) of glyphosate foliar deposits resulted from application of simulated rainfall at 1 hour post-treatment. In combination, these results lend support to the postulate that wash-off with rainfall is a major mechanism of dissipation for glyphosate foliar residues where rainfall occurs shortly after application. Within the plant, glyphosate is not metabolized (Gottrup et al, 1976; Wyrill and Burnside, 1976) but is very well translocated with photosynthates in the phloem. Thus glyphosate is a highly effective compound for post-emergent weed control and particularly for species which resprout from rhizomes or roots. Thompson et al. (1994) quantified initial deposits and subsequent dissipation of glyphosate residues in sugar maple (*Acer saccharum* Marsh.) foliage following ground-based applications of VISION (formulation equivalent to ROUNDUP). Maximum initial residues were 529 mg/kg a.i. dry mass and were highly dependent upon application rate (r^2=0.63-0.87) increasing by a factor of (233-313 mg kg^{-1}) for each kg/ha applied. Foliar residues dissipated exponentially with time, with mean DT_{50} values approximating 2 days, and declining to less than 10% of initial values within 24 days of application. Deposition and persistence of glyphosate in forest plants following aerial applications have been reported by a number of researchers (Newton et al. 1984; Feng and Thompson 1990; Ernst et al. 1987; Thompson et al. 1997) with maximal residue concentrations ranging from 5 to 489 mg/kg. Studies by Newton et al. (1984) as well as Feng and Thompson (1990) are most pertinent to this assessment and show initial foliar residues of glyphosate ranging from 89 to 489 mg/kg for salmonberry and red alder. Residues of AMPA were less than 2% of coincident glyphosate residues for both foliage types. Newton et al. (1984) observed a rapid dissipation of glyphosate residues in hardwood foliage following a rainfall 12 h after application, again reflecting susceptibility of foliar residues to rain wash. Legris and Couture (1989) examined foliar residues resulting from ground applictions of glyphosate at rates equivalent to 1.4 kg/ha and observed maximal foliar residues of 45.3 mg/kg in raspberry foliage which degraded rapidly (DT_{50} < 27 days). Studies by Siltanen et al (1981) and Roy et al (1989) have examined the fate of glyphosate in wild berries of forest environments. The latter study showed that less than 10% of glyphosate penetrated fruit in the first 9 hrs post-application. A gradual decline in residues was observed with estimated DT_{50} of 8 to 26 days and 6 to 14 days for blueberry and raspberry respectively.

The biological significance of such vegetative residues is difficult to assess, however under the assumptions as decribed in section 6.3, maximal vegetative residues expected within the AOC can b e calculated as (313 mg/kg per kg/ha applied x 0.64 kg/ha = 200 mg/kg). This estimate approximates the midpoint or foliar residues actually observed under several field study scenarios as described above, and are markedly lower than LC_{50} values of > 4640 mg/kg as estimated for 8-day feeding studies in bobwhite quail and mallard ducks (Sassman et al. 1984). Evans and Batty (1986) also concluded that glyphosate was non-toxic or only slightly toxic to zebra finches which were allowed unrestricted access to high concentrations in seed. Hoffman and Albers (1984) reported on potential embryotoxicity and teratogenicity of 42 herbicides, insecticides and petroleum contaminants to mallard eggs, found 2,4-D, glyphosate, and atrazine to be only slightly or nontoxic (i.e. LC_{50} values equivalent to 196 to 550 kg/ha). It is generally accepted that direct herbicide toxicity to birds foraging in treated sites is unlikely and that while indirect effects through habitat alteration are more likely, they are relatively unimportant relative to those of other silvicultural practices such as prescribed burning (Millikin, 1994). This perspective is supported by studies of Woodcock et al. (1997), Morrison and Meslow (1983, 1984a and b) and Freedman et al (1988). Woodcock et al. (1997) recently examined the effects of alternative conifer release treatments on forest songbirds demonstrating negligible reductions in bird numbers and that herbicide treatments including VISION (formulation equivalent to ROUNDUP) had no direct effect on most breeding songbirds one growing season after treatment. In similar studies, Freedman et al. (1988) had previously equivalent findings. Morrison and Mellow (1984), examined habitat and bird nesting in response to changes in vegetation on 2 clearcuts receiving aerial applications of glyphosate in western Oregon. Although no difference in overall bird densities were observed in any year of study, individual species densities and habitat usage in year 1 post-treatment. By 2 years post-treatment many species returned to normal habitat usage patterns. Conflicting results have been reported by Cayfor (1988) and Eggestad et al. (1988) for studies of glyphosate applications on black grouse habitat, the former demonstrating a significant preference amongst adults in summer use of treated areas, while the latter indicated a statistically insignificant effect of avoidance of treated areas, particularly for cock birds.

In soils, glyphosate is principally and rapidly degraded by microbial activity (Rueppel et al., 1977) with the primary metabolite being aminomethyl phosphonic acid (AMPA). AMPA is known to be more resistant to microbial degradation than the parent compound (Rueppel et al. 1977) and thus it is somewhat more persistent in soils. A number of studies have examined the of glyphosate in forest soils (Feng and Thompson, 1989; Roy *et al.* 1989, Newton et al. 1984; Henonen-Tanski 1989; Torstensson et al. 1989) and all show that glyphosate is neither persistent in nor susceptible to leaching from this ecosystem compartment. The study of Henonen-Tanski (1989) showed a temperature-dependent effect on soil degradation of glyphosate, with 10 °C reductions in temperature reducing degradation rates by 1/10 or 1/15th of values for higher temperature treatments. DT_{50} values (10 to 14 days) for glyphosate residues in litter of alder and salmonberry as determined by Newton et al. (1984) were similar to those found by Feng and Thompson (1990). In the latter study, > 90% of glyphosate residues were retained in the 0-15 cm organic soil layer and a similar pattern was observed for AMPA. Recently, Thompson et al (1998) examined the fate of glyphosate residues in litter and humus layers following ground applications to a regeneration site dominated by red maple. Again, glyphosate residues in both layers dissipated rapidly following curvilinear kinetics with overall mean DT_{50} values of 10.3 ± 1.24 and 12.0 ± 0.85 days, respectively, with no evidence of leaching. Several studies have shown that glyphosate residues in soils do not adversely effect soil micro-organisms or their metabolic processes (Muller et al. 1981; Fletcher and Freedman (1986). In the study by Fletcher and Freedman (1986), laboratory studies with two leaf litter and one forest floor substrate showed that for herbicides 2,4-D, 2,4,5-T or glyphosate, residue concentrations greater than 50x those of normal field levels are required to reduce litter decomposition rates.

Owing to strong its strong ionic-binding characteristics (Helling, 1971; Sprankle *et al.*, 1975a,b, Rueppel *et al.*, 1977), glyphosate is essentially immobile in soils (Comes *et al.*, 1976; Edwards *et al.*, 1980; Torstensson, 1985; Bowmer *et al.*, 1986, Feng and Thompson, 1989; Roy *et al.* 1989) As a result, the potential for mobilization of glyphosate from treated forest soils is minimal, except in situations where significant storm events occur within the first 2-3 days post application. Studies by Edwards *et al.*, (1980) document the potential losses of glyphosate from vegetated watersheds of slope 6-16% and suggest maximal runoff totalling 1.85% of applied glyphosate. Essentially all of this material was mobilized in the first rainfall event 1 day post-application. At an application rate of 1.10 kg/ha (1lb acre) which most closely approximates that used in the AOC (0.64 kg/ha), Edwards and co-workers reported maximal runoff concentrations of 0.94 mg/L. Comes et al. (1976) examined the desorption of glyphosate residues applied to dry ditches showing that neither glyphosate nor AMPA were detectable in the first water flow through canals where underlying soils contained concentrations of 0.35 and 0.78 mg/kg respectively. The fate of glyphosate in aquatic environments has also been extensively studied. Collectively, results demonstrate that glyphosate is non-persistent in aqueous phase with DT_{50} values typically being < 2 days in ponds or lakes (Legris *et al.* 1987, Legris and Couture, 1989; Legris and Couture, 1990; Beck (1987)). Goldsborough and Beck (1989) conducted studies in four small boreal forest ponds (pH=6.97-8.10) as well as in situ microcosms. Glyphosate application at an intermediate rate (0.89 kg a.i./ha) resulted in aqueous DT_{50} for glyphosate residues of 1.5-3.5 days, with residues falling below detectable levels (0.0005 mg/L) within 37 days in all ponds. AMPA was seldom detected and had maximal concentrations in water less than 0.002 mg/L.). Similarly, glyphosate residues dissipate rapidly from flowing water systems, reaching non-detectable levels within 96 hrs (Feng et al. 1990). Results of several experiments also show that glyphosate has a tendency to sorb to suspended sediments, bottom sediments and biofilms (Feng et al. 1990; Comes et al., 1976; Rueppel et al., 1977; Edwards et al., 1980; Ghassemi et al., 1981; Norris et al., 1983; Newton et al., 1984; Wan, 1988; Bowmer et al. 1986). Sediment residue concentrations vary depending upon the system under study but are relatively persistent. For example residues of 0.25 to 1.45 mg/kg remained 27 months following application in the study by Legris and Couture (1990). Similarly Newton et al. (1984) found stream sediments accumulated glyphosate to a maximal concentration of 0.50 mg /L at approximately 14 d after application and dissipated slowly over the 55 day period of observation. Bowmer et al. (1986) has shown that 13-27% of aqueous phase glyphosate was attenuated for each km of downstream travel and that overall more than 63% of glyphosate was attenuated by sorption to bottom and suspended sediments. Following application to dry sediments, only 7% was redissolved in water.

As observed for atrazine, and consistent with the phytotoxic mode of action for these compounds, plants are equally or more sensitive to aqueous residues of glyphosate than other types of biota (Roshon et al. 1998). Duckweed (*Lemna spp.*), a simple floating vascular plant, appears to be substantially more sensitive to glyphosate than most other aquatic plant species, significant growth reductions being commonly reported at concentrations less than 10 mg/L (Gianfagna and Foy 1975; Cooley and Foy 1986; Prasad 1984; Hartman and Martin 1985). Perkins (1997) and Roshon (1997) have reported EC50 values of 28.8 and 2.98 mg/L a.i. for formulated and technical glyphosate respectively in a representative freshwater macrophyte - *Myriophyllum spicatum.* These values are generally equivalent to LC_{50} values reported for most sensitive fish species (Table 7). Due to the use of glyphosate as a herbicide to control nuisance aquatic vegetation, effects on aquatic plants are well documented. In general, results show that high rates of application (2-6 kg/ha) and correspondingly high aqueous concentrations are required to effectively control aquatic macrophytes (Singh and Muller, 1979; Evans, 1978; Peverly and Crawford, 1975). Further, re-infestation from seed sources often results in rapid recolonization. Sullivan *et al.* (1981) have observed increased numbers of diatoms *Tabellaria*, spp., *Navicula* spp., and *Cymbella* spp. in the sediments of a pond treated with 2.2 kg/ha, which probably represent indirect effects of eradication of emergent vegetation and resultant higher light and nutrient levels. Goldsborough and Brown (1988) reported EC_{50} concentrations for carbon assimilation in periphyton ranging from 9.7 to 35.4 mg/L. A small forest stream in British Columbia was treated with glyphosate (ROUNDUP) at ~ 2.2 and 22 kg/ha, sprayed on the stream surface from a hand-held boom. Sullivan *et al.* (1981) monitored the diatom populations (attached algae) of the treated stream sections and a nearby reference site, and were unable to attribute changes in diatom populations at the sites to effects of the herbicide treatment. Ernst *et al.* (1987) sampled the periphyton of a forest stream in Nova Scotia that was oversprayed (double swath) by helicopter with Roundup at 2.0 Kg/ha. The maximum concentration of glyphosate in the stream was 0.039 mg/L. There were no changes in periphyton biomass (dry weight), chlorophyl *a* content, or ATP production after the application. Glyphosate (Vision) was added to outdoor stream channels at concentrations of 0.001 – 0.287 mg/L and the effects of the herbicide on periphyton were determined (Austin *et al.* 1991). There were no changes in periphyton community structure, but the herbicide treatments resulted in increases in periphyton biomass. They concluded that this was a eutrophication effect of increased phosphorous. Holtby and Baillie (1989) measured periphyton biomass in stream sites oversprayed or adjacent to aerial applications of Roundup at 2.0 Kg/ha in British Columbia. They reported some suggestion of toxic effects on periphyton over the short-term (1 month), but that in the following year, periphyton biomass was higher in treated sites than in reference streams. They also attributed this to an increase in the available phosphorous after the glyphosate applications. In a concurrent experiment, Holtby and Baillie (1989) measured the effects of the Roundup applications (2.0 Kg/ha) on the decomposition of organic debris (macroinvertebrate and microbial activity) in an oversprayed stream. They found that decomposition rates increased in the treated stream for three summers after the applications, and suggested that this was a secondary effect of temperature increases resulting from reduced riparian vegetation. The macroinvertebrate abundance and composition at these sites did not indicate direct effects on the invertebrate community for 22 months after the applications. Kreutzweiser *et al.* (1989) monitored the invertebrate drift at sites oversprayed or adjacent to aerial applications in the same experimental area, and found that at stream sites where maximum glyphosate concentrations were near 0.11 to 0.16 mg/L, there were slight and transient increases in the drift of two macroinvertebrate species. Scrivener and Carruthers (1989) measured macroinvertebrate densities in the substrates of streams and swamps oversprayed with Roundup at 2 kg/ha and compared them to nearby reference sites and found no significant differences in macroinvertebrate densities at treated and reference sites. Among fish, salmonids tend to be the more sensitive to glyphosate intoxication than other fish species (Table 7). Laboratory studies document that the formulated product ROUNDUP is substantially more toxic than the technical product and this has been attributed to toxicity induced by the surfactant (MON0818) contained in the commercial formulation, lowest LC_{50} values being approximately 1-2 mg/L (Folmer et al. 1979; Servizi et al. 1987; Wan et al. 1988). It should be noted, that these values are generated from studies in which exposure levels are held constant for a period of 96 h. However, in natural flowing water systems pulse-exposures much shorter in duration are commonly observed (Wan et al, 1988, Newton et al. 1978, Feng et al. 1990, Thompson et al. 1991, 1995; Kreutzweiser et al. 1995; Newton et al. 1984) period. Folmer and co-workers (1979), conducted tests designed to simulate actual field exposures in which sac-fry of rainbow trout were exposed to ROUNDUP for 6 hrs. Results of these tests indicated that statistically significant reduction in survival occurred only at concentrations equal or greater than 5 mg/L. Janz et al. (1991) reported on threshold herbicide concentrations causing physiological stress in juvenile coho salmon. These researchers exposed fish for short periods (4 hrs) to forest-use herbicides and showed that glyphosate (as IPA) toxicity thresholds under this exposure regime exceeded 96 hr LC_{50} values, suggesting no significant physiological stress response in this species on short-term exposures. Mitchell et al. (1987), Chapman (1989), as well as Folmar (1976) conducted laboratory toxicity studies on an variety of salmonid species and concluded that at recommended use-rates, ROUNDUP posed no acute toxicity hazard to salmonids. This conclusion has been supported by a number of subsequent field studies involving exposure of salmonids under operational or semi-operational scenarios, all of which show no significant mortality to salmonids (Hildebrand et al 1982; Holtby and Baillie 1989). The effects of glyphosate on fish were determined in a forest stream in British Columbia that was hand-sprayed at rates of 2.2, 22.0 and 220 Kg/ha (Hildebrand *et al.* 1982). There was no mortality and no indication of unusual behaviour of rainbow trout fingerlings placed in cages in the treated sections of the stream. These authors also monitored the survival and behaviour of caged rainbow trout in a stream that was oversprayed by helicopter at 2.2 Kg/ha, and reported no mortality or behavioural effects. Holtby and Baillie (1989b) placed caged coho salmon fingerlings in treated streams (oversprayed or adjacent to aerial applications of 2.0 Kg/ha Roundup) and reference streams, and monitored survival and behaviour. They observed some signs of stress in coho fingerlings at the oversprayed sites (apparent increased respiration) at about 2 h after the application, and found 2 - 4% mortality in cages at the treated sites within 12 h post-application. Indications of treatment effects on survival were confounded by 8% mortality in the reference cages by 24 h after the applications. These authors also monitored the density, movement and growth of coho fingerlings in an oversprayed tributary for 2 yrs. Their data suggested a temporary (3 wk) reduction in fish movement within the tributary shortly after the applications, which they submitted was indicative of sublethal effects causing stress. Apart from this apparent short-term effect, there were no measurable changes attributable to the herbicide applications in growth rates, movement, or over-winter mortality in the 2-yr post-treatment period. Newton *et al.* (1984), reported on a comprehensive whole watershed fate study, conducted in Oregon, in which glyphosate was applied aerially at a rate of ~ 3 .3 kg/ha. The maximum concentration observed in the oversprayed stream was 0.27 mg /L with residues decreasing rapidly to less than 0.1 mg by 4 hrs after application. Coho salmon fingerlings placed in the lower portion of the treated stream did not accumulate detectable levels of glyphosate.

Perkins (1997) examined the effects of both ROUNDUP and RODEO formulations of glyphosate on frogs (*Xenopus laevis*), reporting widely differing LC_{50} values of 9 and 5407 mg/L respectively. The difference in toxicity estimates were attributed to the influence of the toxic surfactant in ROUNDUP which is absent from the RODEO formulation. The LC_{50} for the MON0818 surfactant itself was reportedly 2.7 mg/L. Perkins (1997) cited unpublished data of Berrill et al. (1995) who examined the effects of ROUNDUP and RODEO on several different frog species including the wood frog (*Rana* *sylvatica*), the leopard frog (*R. pipiens*), the green frog (*R. clamitans*), the bullfrog (*R. catesbeiana*) as well as the American toad (*Bufo americanus*). Both embryos and tadpoles of theses species were exposed to 6, 7 and 8 mg/L concentrations of the two glyphosate formulations for 96 h. At 8 mg/L 17-100% of tadpoles exposed to ROUNDUP died or showed abnormalities, while at the same concentration of RODEO no effects were observed.

It must be noted here that many laboratory studies involve exposure regimes and conditions that differ substantially from those expected in natural environments. For example glyphosate in aquatic systems is known to partition preferentially to sediments to which it is strongly bound and from which it is poorly desorbed, but this potential is rarely incorporated into laboratory testing protocols. Limited studies (Paulson *et al.*, 1975; Sutherland *et al.*, 1976; Khan *et al.*, 1985; Khan 1991) suggest that herbicide residues bound to soils and plants are generally not biologically available. A second example involves the *Xenopus laevis* test protocol in which amphibian eggs are "de-jelled" by chemical treatment prior to exposure to herbicides, thus eliminating a natural protective barrier and maximizing the potential for toxic effect.

6.5 Environmental Chemistry, Fate and Toxicology of Triclopyr

Triclopyr, is a chlorinated pyridine derivative, discovered and developed originally by the Dow Chemical Company (now Dow AgroSciences), and registered in 1982. Two formulations of triclopyr are available for use in forestry. The ester formulation marketed in the U.S.A under the trade name GARLON 4 (and in Canada as RELEASE) contains the formulated ingredient triclopyr butoxyethyl ester (TBEE), while the amine formulation, marketed in both countries as GARLON 3A contains the triethylamine salt (TEA) of triclopyr. The amine formulation is employed principally for application to cut-surfaces of plants, while the ester formulation with greater cuticular penetration characteristics, may be applied to foliage or as a basal bark treatment. A detailed review of triclopyr in relation to forest uses (USDA-FS, 1984) documents typical application rates for the ester formulation ranging from 0.5 to 5.8 kg/ha., while use rates for the amine formulation range from 1.4 to 8.1 kg/ha. Triclopyr is a broad-spectrum herbicide originally developed for vegetation control on rights-of-way and industrial sites (McKellar et al. 1982) and provides good control of most woody plants in forest site preparation scenarios and for conifer release in certain more tolerant crop types such as black spruce (Campbell 1991). The mode of action of triclopyr is imperfectly understood but very similar to that of 2,4-D as described below. As shown in Table 4, triclopyr ester is characterised by lower water solubility and relatively high K_{ow} and K_{oc} values as compared to the amine and acid forms. Predictably, esters therefore penetrate plant cuticles more effectively and sorb or concentrate in either biotic or abiotic organic matrices to a greater extent than acid or amine forms. TBEE hydrolysis in water has been shown to be base-catalyzed and proportional to temperature, with *DT_{50}(hydrolysis)* at 25 °C estimated as 84, 8.7 and 0.5 days at pH 5, 7, and 9 respectively, and 208, 25.5 and 1.7 days at 15 °C (McCall *et al.* 1988). As with most salts, TEA dissociates rapidly to form the acid upon release into the environment. In contrast the acid form is relatively susceptible to hydrolysis (Ghassemi *et al.* 1981) at environmentally relevant pH and temperatures regimes but is susceptible to photolytic and microbial degradation. The principal metabolite of triclopyr is 3,5,6-trichloropyridinol (TCP) which is itself extremely photolabile and thus rarely detected in the environment.

In plants, triclopyr is taken up by both roots and foliage and is readily translocated with photosynthates throughout the plant system. Translocation rates are temperature and photoperiod dependent (Radosevich and Bayer 1979) with maximal rates occurring under warmer, longer day-length conditions. King and Radosevich (1979) reported greater uptake in immature as compared to mature leaves of tanoak, probably reflecting age differentials in cuticular development. Bentson (1990) has recently reviewed the fate of xenobiotics in foliar pesticide deposits noting that the topic area has received little holistic study and that the major dissipative processes involved are wash-off by rain, penetration (and translocation), volatilization and photodegradation. Most recently, Bentson and Norris (1991) have investigated the influence of temperature, illumination and time on the disposition of triclopyr residues in pacific madrone (*Arbutus menziesii*) and giant chinkapin (*Castanopsis chrysophylla*) following application of TBEE. These studies documented an exponential increase in loss of TBEE with increased temperature and significant losses in foliar deposits via photolysis. In both cases, the effects were species specific, indicating that degradative losses were more important in species with less penetrable surfaces, while uptake and translocation are the dominant processes when penetration of the leaf cuticle is facile. Thompson et al. (1994) documented foliar residues of triclopyr following ground-based applications to sugar maple. Maximum initial residues were 1630 mg/kg dry mass with foliar residues increasing proportionally by a factor of 233-313 mg/kg for each kg/ha applied. Foliar residues dissipated exponentially with time, with mean DT_{50} calculated as 1.5 days for TBEE and 4 days for the acid form. Triclopyr acid residues declined to less than 10% of initial values within 33 days of application. Based on the maximal residue levels observed in this study, the authors estimated a dose level of ~ 408 mg kg^{-1} b.w. for birds which was substantially lower than the lowest acute oral toxicity values reported by Sassman et al. (1984) for triclopyr acid (1,698 mg kg^{-1} b.w. for mallard ducks). Studies by Holmes et al. (1994) examining lethal and sublethal effects on caged zebra finch also suggested that environmentally realistic exposures of TBEE expected in conjunction with forestry applications should not induce deleterious effects on small birds. Woodcock et al. (1997) recently examined the effects of alternative conifer release treatments on forest songbirds demonstrating negligible reductions in bird numbers and that herbicide treatments including RELEASE (formulation equivalent to GARLON 4) had no direct effect on most breeding songbirds one growing season after treatment.

In soils, triclopyr is moderately sorbed and non-persistent, being degraded by both chemical and microbial mechanisms. Although there is no direct evidence of photodegradation of triclopyr on soil surfaces, rapid photolysis in both water and on leaf surfaces (Bentson 1991) suggests that it is likely. McKeller et al (1982) estimated the DT_{50} for triclopyr to be 14-16 days in a West Virginia watershed. The Dow Chemical Company (1983 a) reports various DT_{50} estimates depending upon soil type ranging from a low of 10 days in a silt clay loam to 46 days in a loam soil. In Swedish forest soils low concentrations of triclopyr (~ 10% of applied) remained in excess of 2 years post application in some soil types. Norris et al (1987) examined the dissipation of triclopyr in two pasture situations in Oregon and estimated DT_{50} values of ~ 80 days, with only trace levels of TCP detected. Thompson et al. (1998) examined triclopyr persistence in litter and humus soils of eastern Canadian forests following ground-based applications and observed mean maximal residue levels of 54.19 mg/kg in litter and 5.02 mg/kg in organic soil 35 days post-application. Although triclopyr residue levels were comparatively more variable and persistent than those for glyphosate , a general slow linear decline with time was observed such that less than 10% of mean maximal residues remained by end of the 77 day observation period. The fact that residue levels were approximately 10-fold higher in litter than in organic soil layers suggested a low propensity to leach under these conditions. Data from Stephenson et al. (1990) showed equivalent DT_{50} values of ~14 days in both sandy and clay soils of the boreal forest. While some evidence of leaching was observed in response to a heavy rainfall event 7 days after application, more than 97% of triclopyr was recover within the upper 15 cm layer. This result was consistent with findings of Lee et al (1985) who studied the leaching behaviour of triclopyr in laboratory soil columns and found that all residue remained in the top 10 cm of the soil. Maximum concentrations observed in a drainage ditch downslope of small plots (6-7% slope) treated at rates of 2.6 to 3 kg/ha a.i., contained maximal concentration of 0.001 mg/L. In other studies examining runoff losses of triclopyr from treated watersheds similarly little evidence of runoff losses have been observed. The study of Thompson et al (1991) indicated that relatively small amounts of triclopyr residues may be mobilized from treated surfaces in association with rainfall events occurring soon after the chemical is applied, and that mobilization is insignificant in later events. These studies support the findings of Norris and co-workers who suggested that neither long-distance overland flow nor leaching introduced significant amounts of triclopyr into a stream draining a treated watershed. Further, residues generated via runoff would be expected to contain proportionally less of the more toxic ester form of the compound (Thompson et al. 1991).

In aquatic ecosystems, the fate of TBEE as the more toxic chemical moiety, is of critical interest. A small forest stream in British Columbia was treated with triclopyr (Garlon 4) by direct application to the stream surface with a hand-held boom at a rate of 7.7 kg/ha a.i. (Mayes *et al.* 1989). Triclopyr residues in stream water peaked at 1.9 mg/L (total triclopyr) shortly after the application, and declined to 0.23 mg/L within 8 h. Caged coho salmon and rainbow trout fingerlings, mayfly nymphs (*Ameletus* sp.), and stonefly nymphs (*Acroneuria* sp.) were placed in treated and reference sections of the stream. By 4 d after the application, there was no mortality of coho salmon, 13% mortality of rainbow trout in treated sections (none in controls), and no mortality attributable to the herbicide of either insect species. Thompson et al. (1991) documented the fate of TBEE following direct aerial application to a boreal forest stream. The average deposit monitored at the stream surface was 3.67 kg/ha a.e. (range = 3.35 to 3.99 kg/ha a.e.). Streamwater residues of TBEE resulting from direct overspray were characterized by instantaneous maxima (0.23 to 0.35 mg/L) and a series of diminishing pulses of chemical associated with inputs upstream of the sampling site. TBEE concentrations declined to levels below the limits of quantification (0.001 mg/L) within 72 hours post-application. Transient residues of the acid were observed in streamwater, with a maximum concentration (0.14 mg/L) 6 hours post-application but TCP residues did not exceed limits of quantification (0.05 mg/L) in any sample. Results indicate that natural dissipation mechanisms reduce both the period and concentrations to which aquatic organisms would be exposed to triclopyr residues in natural stream systems. In association with this study (Fontaine, 1990) studied survival of caged yellow perch, fathead minnows, caddisfly larvae (*Hydropsyche* sp.) and crayfish in treated and reference sections of the stream. There were no significant differences in mortality of either fish species between treated and reference sections, although high mortality of fathead minnows at the control site (about 45 %) warrants caution in concluding that there were no effects of triclopyr on the survival of fathead minnows. No significant mortality of caddisfly larvae occurred, but significant mortality of crayfish was recorded by 96-h post-application. Fontaine suggested that because of delayed mortality of crayfish at the treated sites, this mortality was not related to the triclopyr application In a follow-up study conducted in a small first order forest stream, maximal concentrations of 0.848 μg mL^{-1} and 0.949 μg mL^{-1} were observed following direct injection of TBEE to streamwater. Thompson et al. (1995) studied the fate of triclopyr following direct injection into a small first-order stream system. Average TBEE concentrations ranged from 0.32 μg mL^{-1} at stations nearest injection points to 0.02 μg mL^{-1} approximately 225 m downstream. Periods of exposure to TBEE concentrations in excess of 0.001 μg mL^{-1} ranged from 55 min in fast-flowing upstream locations to 120 min at slower, downstream sampling locations. Simultaneous quantitation of triclopyr acid (TRI) residues in stream water samples indicated that natural degradative mechanisms rapidly converted TBEE to TRI, and that sorption to natural allocthanous materials further attenuated TBEE concentrations as chemical pulses moved downstream. High concentrations of TBEE were observed in allocthanous (deciduous leaf) material in this experiment however followup laboratory studies (Kreutzweiser et al. 1998) showed that such high concentration were non-toxic to macroinvertebrates which process this type of material.

Owing to its high polarity, triclopyr acid exhibits a very low potential to bioaccumulate (BCF=0.5 in bluegill sunfish) as compared to the calculated BCF value of 400 for the ester (McCall *et al.* 1988). The difference in bioaccumulation potential is consistent with the substantial difference in K_{ow} (0.205 and 12300, respectively) for the two forms of the compound. The difference in bioaccumulation potential and the related facility of TBEE to penetrate biological membranes is a key factor inducing substantially higher toxicity for ester forms of this compound as compared to acid and amine forms (Table 7). As with many other toxicants, salmonids tend to be more senstive than other fish. Solomon *et al* (1990), estimated a DT_{50} value of 4.3 days for triclopyr acid based on their studies employing in situ enclosures in a boreal lake. In similar studies conducted by Kreutzweiser and co-workers (1995) a DT_{50} value of 3-4 days for TBEE was observed irrespective of initial concentration. During the 15 day study period, essentially all of the triclopyr ester (> 90%) was converted to the free acid. Vertical depth sampling indicated that the herbicide mixed homogeneously within the enclosed water column within 24 hours of application. Again salmonid fishes tend to be particularly susceptible to triclopyr toxicity with lowest 96 hr LC_{50} values ranging from 0.5 to 1.2 mg/L. Janz et al. (1991) have shown that threshold herbicide concentrations causing physiological stress in juvenile coho salmon exposed to either triclopyr (as TBEE), triclopyr(as TEA) or glyphosate (as IPA) for short periods (4 hrs) exceed 96 hr LC_{50} values. This result indicates that neither triclopyr or glyphosate herbicides induce significant physiological stress response in this species on short-term exposures. Kreutzweiser et al. (1994) examined the influence of exposure duration on the toxicity of triclopyr ester in two sensitive salmonid species under flow-through test conditions. Results showed a marked effect of exposure time with 1 h and 24 h exposures yielding LC_{50} values of 22.5 and 0.79 mg/L, and 34.6 and 1.76 mg/L for rainbow trout and chinook salmon respectively. As noted by Kreutzweiser et al. (1995), results of laboratory time-toxicity studies as well as field studies (Thompson et al. 1991, Fontaine, 1990, Thompson et al., 1995, Mayes et al. 1989) suggest that there is little risk of acute lethal effects in fish or other aquatic organisms even following exposure to maximal environmental concentrations in flowing water systems. In contrast, under cool, low-light irradiance, non-flowing conditions where triclopyr may persist for several days in the ester form substantial toxicological effects, particularly on salmonid fish may occur as evidenced by the field studies conducted by Kreutzweiser et al (1995) in which all caged rainbow trout exposed to concentrations above 0.5 mg/L were killed. Triclopyr (as Garlon 4) was applied to outdoor stream channels in northern Ontario at concentrations up to 320 mg/L, and the lethal and behavioural effects of the treatments on five species of aquatic insects were determined (Kreutzweiser *et al.* 1992). The triclopyr applications resulted in significant drift and mortality of the caddisfly *Dolophilodes distinctus* at 3.2 mg/L (no effects at 0.32 mg/L), the stonefly *Isogenoides* sp. at 32 mg/L, and the mayfly *Epeorus vitrea* at 320 mg/L. Significant lethal effects on the caddisfly *Hydropsyche* sp. and behavioural effects on the mayfly *Isonychia* sp. occurred, but only at the maximum test concentration of 320 mg/L. Berrill et al (1994) examined the effects of low concentrations of forest-use pesticides on embryos and tadpoles of several frog species (*Rana catebeiana* (bullfrog), *Rana pipiens* (leopard frog) and *Rana clamitans* (green frog). The authors suggested that the jelly layer surrounding frog eggs provides protection since even the high test concentrations of fenitrothion (insecticide) hexazinone and trilcopyr (applied as the ester) showed no effects on embryo hatching success. Tadpoles of the three species exposed to 0.6 and 1.2 ppm for 48 hrs suffered minimal mortality, however newly hatched tadpoles were highly sensitive to triclopyr at concentrations of 2.4 and 4.8 mg/L either dying or remaining paralyzed following exposure. Tadpoles initially affected by exposure to lower concentrations of triclopyr usually recovered within 1 to 3 days. Bullfrog tadpoles tended to be the most sensitive test species. Perkins (1997) examined the effects of both GARLON 4 and GARLON 3A formulations on frogs (*Xenopus laevis*), reporting differing 96 h LC_{50} values of 9 and 163 mg/L respectively. The difference in toxicity estimates were attributed to the lipophilic nature of the ester form of triclopyr in GARLON 4, which is more rapidly taken up and accumulated in biological systems (McCall et al.1988; Barron et al. 1990).

6.6 Environmental Fate and Persistence of Sulfometuron-methyl

Sulfometuron-methyl is a member of the relatively new class of sulfonylurea herbicides introduced in 1975 and registered by E.I. DuPont de Nemours Inc in the early 1980s. Sulfonylurea compounds including sulfometuron methyl, are extremely potent inhibitors of the acetolactate synthetase enzyme in plants and thus are applied at very low use rates (< 0. 2 kg/ha a.i.) which is generally an order of magnitude lower than for conventional herbicides. Average use rates for this compound within the AOC (0.17 kg/ha a.i.) are typical for the product. Sulfometuron-methyl is used in both pre- and post-emergent weed control programs and is particularly effective on grasses. The environmental fate of sulfonylurea herbicides including sulfometuron-methyl was reviewed by Blair and Martin (1988). Sulfometuron is characterised by a moderate water solubility and Koc values and a low Kow. As such it is moderately susceptible to leaching and not bioaccumulatory. Sulfometuron-methyl is degraded principally by hydrolytic and microbial processes in the environment that yield saccharin as the principal metabolite.

In plants, sulfometuron-methyl is rapidly taken up by both roots and foliage of plants and is readily translocated in the xylem. Michael et al. (1987) examine the environmental fate of sulfometuron methyl following various applications to forests in Mississippi and Florida. They reported that sulfometuron methyl residues maximally average 26.3 mg/kg among several competitive plant species and dissipated rapidly from both the pine crop and weedy vegetation with DT_{50} values of less than 1 day in Florida and 1-3 days in Mississippi, being non-detectable within 60-90 and 27 days at the two sites respectively. The 8 day median lethal concentration for sulfometuron in feed is > 5000 mg/kg for mallard ducks a value which is approximately 2 orders of magnitude greater than residues calculated or observed in forest vegetation, indicating very low potential for effects in birds.

Beyer et al. (1987) have reviewed the behaviour of sulfonylurea herbicides, including sulfometuron-methyl in soils, indicating that both chemical hydrolysis and microbial breakdown are the principal modes of degradation. Degradation is generally fastest in warm, moist, light-textured, low pH soils. In soils and water, sulfometuron-methyl is degraded by both hydrolysis and microbial activity with an average DT_{50} value of 4 weeks in acidic soils (Harvey et al., 1985). The principal non-volatile degradation product is saccharin (Anderson and Dulka, 1985). Among a variety of other soils, the latter authors examined the fate of sulfometuron methyl in a field study in an Oregon sandy loam (OM 1.9%; pH 5.3) and provided data which show an approximate DT_{50} of 112 days. In this soil, the majority of the applied material was found in the upper 0-8 cm layer with < 5% in the lower 24-32 cm layer, suggesting little potential for leaching. Among the five field studies conducted, maximal leaching was observed in Colorado, where the loam soil was characterised by cool temperatures (50 °C), high rainfall (16 inches), and high soil pH (7.3). In the study by Michael et al (1987), dissipation of sulfometuron methyl from litter and soil at the Florida site was rapid with DT_{50} values of 1.8 to 7.2 days for litter and mineral soil respectively. At the Mississippi site dissipation was slightly slower with DT_{50} values for litter and mineral soil of 3.6 to 13.1 days. Sulfometuron residues did not leach below 30 cm, did not persist beyond 2 months and did not contaminate shallow surface -aquifers at the Florida site at levels above analytical detection limits. Anderson and Dulka (1985) noted that soil pH and moisture content are key determinants of overall persistence in soils. Unlike the situation for most herbicides, soil organic matter and clay content are relatively unimportant factors in sulfometuron adsorption (Wehtje et al. 1987). Thin-layer soil chromatography studies indicate that sulfometuron was slightly more mobile than imazapyr. Because hydrolysis is a primary degradation mechanism and sulfometuron-methyl is a weak acid with a pKa of 5.2, it may exist in various ratios of ionic state depending upon soil pH which thus significantly influences sorption, microbial degradation and hydrolysis rates. Hydrolysis rates are also highly dependant upon temperature (Beyer *et al.* 1988). Such effects were evident in the study by Lym and Swenson (1991) who found sulfometuron-methyl to be relatively persistent in silty clay loam (pH 6.1, 8 °C, 45% field capacity) but persisted only 218 d in the same soil at 90% field capacity and 16 °C. Degradation was slower in sandy loam soils with a pH of 7.4 and averaged >700 d, regardless of environmental conditions. Lym and Swenson (1991) also showed that sulfometuron-methyl did not move appreciably from treated slopes of 6-18% but that under certain conditions it was susceptible to leaching. Emphasizing the importance of rainfall intensity and duration, their studies showed sulfometuron moved beyond 70 cm depth in loam, silty clay loam and stony loam soils when leached with 45.7 cm of water for 48 h compared to only 35 to 50 cm deep when leached with the same amount of water over 9 wk. Sulfometuron degradation increased as soil temperature and moisture increased.

Laboratory studies have demonstrated that sulfometuron may be bacterio- and fungi-static to some strains of bacteria in pure but not mixed culture (Scheel and Casida, 1985; LaRossa and Schloss, 1984; Balco and Dumas). Soil residues of sulfometuron methyl are highly phytotoxic as evidenced by several plant bioassays for sulfonylureas which detect effects at levels below 1 ng/g (Zhanow 1982, Joshi et al. 1985). In laboratory studies, Hubbard et al. (1989) studied the surface runoff and leaching behaviour of sulfometuron-methyl in three different soil types and found that surface runoff increased with higher intensity of rainfall and was greatest in sandy clay loam (maximum 34.7% of applied). In contrast, leaching was greatest in sandy loam soils (~82 % of applied) with only minor differences in relation to rainfall intensity. The authors suggested that in most cases percolation will be the major loss pathway for sulfometuron-methyl on sandy soils, whereas runoff will predominate on clay soils. Beyer et al. (1987) noted that sulfonylureas are characterized as relatively mobile compounds and that depending upon rainfall, net soil water movement and degree of soil drainage, this mobility can be important. Stone et al. (1993) investigated the leaching potential of sulfometuron methyl in soil columns with and without forest litter and humus material and found that this compound dissipated within 80 days in the lab environment and did not leach beyond 20 cm soil depth. Wauchope et al (1990) investigated the effects of application rates, grass cover, and formulation type on herbicide losses in runoff, folllowing applications of cyanazine (4.5 kg/ha) or sulfometuron methyl (0.4 kg/ha) to bare-soil or grass-covered plots(3% slope) of a loamy sand soil. Following herbicide applications, simulated rainfall events of 69 mm/h intensity until 2 mm of runoff was applied and runoff was was analyzed for sediment and herbicide concentrations. The bare plots required one-third less rain to produce the same amount of runoff and yielded twice as much sediment as the grassy plots. However, losses of all formulations were 1 to 2% of the amounts applied regardless of grass cover and even though cyanazine rates were 11 times that of sulfometuron methyl. Total losses of all formulations were sensitive to the length of time between rainfall initiation and runoff initiation, indicating that leaching made herbicide unavailable for runoff. These results suggest that, for these formulations under conditions of similar runoff volumes, losses of pesticides are a fairly constant fraction of the amounts applied, with or without grass cover. For intense storms where the amount of rainfall is similar, chemical runoff from the grassed plots was predicted by computer simulation to be less than half of that from bare soil. In combination results of these studies by (Wauchope et al. 1990, Michael 1987 ; Lym and Swenson, 1991; Wehtje et al. 1987) support the contention of the latter authors that at rates < 0.13 kg/ha and in soils less than pH 7, sulfometuron methyl has little potential for movement into groundwater but that considerable movement may occur in saturated soils with pH > 7 and where intense rainfall events occur over short time spans.

In water, sulfometuron-methyl is susceptible to both hydrolysis and photolysis. Lym and Swenson (1991) found sulfometuron hydrolysis was independent of solution pH (5 to 9) with an average of 63% 14C-sulfometuron remaining after 28 days. These findings conflicted with those of Harvey et al. (1985) who observed a marked influence of pH on aqueous stability. The difference in average DT_{50} values for ultraviolet-irradiated samples (31 days) and dark control samples (65 days) demonstrated that sulfometuron is susceptible to photodegradation. Neary and Michael (1989) examined the effect of sulfometuron methyl on groundwater and stream quality in coastal plain forest watersheds of Florida and Mississippi. In the former site, following ground applications of a pelleted formulation at a rate of 0.39 kg/ha, maximal residues of 0.007 mg/L were observed in a stream protected by a 5 meter buffer strip with no-detectable residues beyond 7 days. Residues in streamwater were non-detectable at distance of more than 150 m downstream. At the Mississippi site a large buffer zone was employed in conjunction with the aerial application. Maximal streamwater concentrations under this scenario were 0.044 mg/L with residues at or below non-detectable levels beyond 29 days post-application. As typical, most of the offsite movement occurred with the first 2 to 3 storm event at both sites. No sulfometuron residues were detected in suspended or bottom sediments at either site. Maximal streamwater residues observed in these studies were more than 200 times less than LC_{50} values reported for aquatic invertebrates and fish (Table 7), suggesting a signficant margin of safety for this compound in relation to potential effects on most aquatic organisms. A NOEC estimate for effects on eggs and larvae of the the fathead minnow has been estimated at 1.2 mg/L, which is approximately 25 times higher than maximal levels recorded in streams. Recently, Fort (1998) reported on the effects of sulfometuron on frog (*Xenopus laevis*)development and metamorphosis reporting 4 day LC_{50} of > 10.0 mg/L and EC_{50} for organogenesis malformation of 0.94 mg/L. Results of 30 day organogenesis test provided estimates of NOEC and LOEC equivalent to 0.01 and 0.05 mg/L respectively. However, the most significant effects for sulformeturon methyl were in 14 day tail-resorption studies, where NOEC and LOEC estimates were 0.001 and 0.01 mg/L and ten-fold lower estimates (i.e. NOEC = 0.0001 and LOEC 0.001 mg/L) were calculated for the rate of tail resorption, respectively. While the author noted that ecological significance of these findings are unknown, the results are clearly evidence of potentially high chronic, sublethal toxicity in amphibians and indicate a need for further research in this area.

6.7 Environmental Chemistry, Fate and Toxicology of Hexazinone

Hexazinone is a member of the symmetrical triazine class of herbicides and thus chemically related to atrazine. Its biological activity in plants results from the same photosynthetic inhibition mechanism and it is the most water soluble compound in the triazine family of herbicides. Originally introduced by E.I. DuPont de Nemours Inc. in 1975, it was developed and registered by the EPA in 1984. Hexazinone is used in production of only a few agricultural crops with sufficient tolerance (blueberries, sugarcane, pineapple and alfalfa), but is used as a nonselective herbicide in industrial applications and as a selective herbicide in forestry, principally as a site-preparation tool. Hexazinone is manufactured in both liquid (VELPAR L) and pellet (PRONONE) and although neither are currently used within the AOC, typical use rates for these products range from 0.6 to 3.3 kg/ha a.i.

In plants, although hexazinone has some foliar activity, it is more effective when taken up via the roots, thus it is typically soil-applied at rates which vary with target species and soil type from 0.45 to 12 lbs/acre a.i.. Lowest use rates are employed where coarser textured soils occur and where more susceptible graminaceous species predominate. Differential metabolism among plants is a factor in selective action of this compound (Wood et al. 1997).

In soils, hexazinone is degraded by both hydrolytic and microbial mechanisms and has been classed as a very mobile herbicide based on soil TLC experiments (Rhodes 1980a,b). The combination of high water solubility, low Kow and low soil partition coefficient in soils (Kd=0.2) suggests that hexazinone may have a high potential for leaching (Stone et al. 1993) or offsite movement with surface water. Bouchard and Lavy (1985) studied the adsorption and desorption of hexazinone in forest soil matrices and showed that leaf litter had the greatest sorptivity for hexazinone (Kd=15.09), that organic carbon was a principal determinant in relative adsorption and that hexazinone was readily desorbed from all matrices. The importance of litter and humus layers in binding hexazinone was also demonstrated by Stone et al. (1993) where leaching in soil columns was approximately 3-fold greater under bare soil conditions as compared to that in columns with litter and humus present. Soil pH is known to be an important factor in the adsorption of many triazine herbicides ((Bailey et al 1968, Weber et al. 1966). Helbert (1990) found that hexazinone leached to a meand depth of 65 cm in a well-drained sandy loam after applications at very high rates. Felding (1992) studied the leaching potential of hexazinone and atrazine following annual applications in 7 to 10 year old fir plantations established on clay soils. Drainage water samples taken from the site contained concentrations ranging from <0.0001 to 0.0021 mg/L in one site and 0.0035 to 0.043 mg/L in a second site. These studies showed that the herbicide leached more than 1 meter in one year and was detectable in drainage water more than 2 years after applicaton. Maximal leaching was attributed to heavy rainfalls just after application, however neither the amount or exact timing of rainfall was specified. Other authors have also demonstrated significant leaching potential for hexazinone, resulting in label restrictions to exclude coarse-textured soils in Canada. Feng et al. (1989) showed peak concentrations of 0.205, 0.121 and 0.089 mg of hexazinone in leachate from tubes placed at 30, 55 and 80 cm depths in the soil profile. Feng et al. (1992) studied the dissipation of hexazinone residues following application of a pelleted formulation (PRONONE 10G) to clay soils of northern Alberta. Soil residues declined to 10% of initial levels within one year of application, with detection of metabolites A and B (15 and 30% of hexazinone levels) indicating degradation had occurred in-situ. Two years following application, hexazinone was quantifiable up to 40 cm depth in the soil. Bush et al. (1990) found no groundwater contamination following applications of 2.8 kg/ha where the water table was 2 to 14 meters below the surface. In a Florida site with similar soils treated at a rate of 1.7 kg/ha, hexazinone concentrations in surficial, unconfined ground water ranged from 0.017 to 0.035 mg/L.

The potential for lateral movement into, as well as fate and potential effects of hexazinone in aquatic ecosystems has been extensively studied, particularly in relation to forestry uses. Much of this work has been conducted by Neary and co-workers in southeastern USA and has recently been previously reviewed (Neary et al. 1993). In a study by Miller and Bace (1980) high concentrations of hexazinone in streamwater (2.4 mg/L) were observed as the result of pellets being deposited directly into the stream. These residues declined to 0.10 mg/L within 24 hrs and less than 0.01 mg/L in 10 days. In contrast, where applications do not directly contaminate streams, hexazinone residue levels in streamwater are typically much lower or non-detectable. Neary (1983a) observed no-detectable levels of hexazinone following operational applications of hexazinone in Tennessee. In a study involving four small (1 ha each) watersheds where hexazinone pellets were applied to sandy loam soils at a rate of 1.68 kg/ha, Neary et al (1983) monitored hexazinone concentrations in stormflows of small first-order ephemeral streams in association with 26 storm events. Residues in runoff were maximal in the first storm after application (mean 0.442 mg/L). However, total loss of hexazinone in storm runoff average 0.53% of the applied herbicide, with two major storm events accounting for the majority (59.3%) of total losses. In an Arkansas study involving hexazinone liquid spot applications, where ephemeral channels were not treated, no-detectable concentrations of hexazinone were observed in stormflows, however baseflows carried low levels of hexazinone (< 0.014 mg/L) for over one year post treatment (Bouchard et al. 1985). Similar concentrations (0.006 to 0.036 mg/L) have been measure in stream flow in several sets of spot treatments in Alabama and Georgia (Michael and Neary 1992). A second-order stream below the treated watersheds periodically contained hexazinone residues of < 0.044 mg/L. Letich and Flinn (1983) monitored hexazinone residues in streamwater draining a 46.4 ha catchment in Australia which was aerially treated with hexazinone at a rate of 2 kg/ha. Soils in the catchment were gravelly clay loams with high infiltration capacity. Slopes over 45% of the catchment were greater than 20 degrees. In a total of 69 samples collected hexazinone was detected in only 6 samples with maximal concentrations of 0.004 mg/L. Low residue levels were attributed to the use of 30 m vegetated buffers along the stream and environmental conditions involving low soil moisture and low rainfall post-treatment (89 mm over 9 weeks).

While the emphasis of hexazinone fate studies has been on leaching, runoff and contamination of streamwater, scientific studies have also been conducted in standing water systems with differing

results. Solomon *et al.*, (1990) reported $_{DT50}$ values of 3.8 and 3.7 days, and DT_{95} of 42 and 21 days respectively for two rates (4.0 and 0.4 kg/ha) of hexazinone applied to in-situ enclosures. Initial concentrations approximately equivalent to the nominal values of 0.017 and 0.167 mg/L respectively, declined to below detectable levels 21 and 42 days post application respectively (limits of detection for hexazinone in water were not reported). Enhanced photolysis and inhibited biotransformation might be expected under the conditions of this study relative to typical lentic systems in forested watersheds. Only minimal residues associated with bottom sediments. More rapid dissipation (DT_{50} < 24 hrs) of initially high (0.820 mg/L) residues were observed by Legris and Couture (1987) in a pond directly oversprayed with hexazinone at a nominal rate of 3.6 kg/ha. Minimal sorption to sediments was also reported with residue maxima of 0.172 mg/kg. In a recent study where hexazinone was applied at various concentrations to replicate in situ enclosures, Thompson *et al.* (1992) observed highly persistent aqueous residues of hexazinone (DT_{50}=131-280 d) depending upon initial concentration (1.0 or 10 mg/L). The unexpectedly long persistence observed in this study was attributed to short-day length and low irradiance impairing the primary photolytic dissipation mechanism (Rhodes 1980b). The impact of hexazinone in non-flowing systems has not been extensively studied. Results presented by Solomon *et al.* (1990) indicate that initial concentrations resulting from surface applications at the maximum label rate (4.0 kg/ha) resulted in significant but temporary (4-14 d post-application) depression of dissolved oxygen (reflecting impact on phytoplankton and/or periphyton). Similarly, significant concentration-dependant depression of dissolved oxygen was observed by Thompson *et al.* (1992) who also confirmed that significant hexazinone-induced impacts on phytoplankton biomass *in Cyanophyta, Chlorophyta, Chrysophta, Cryptophyta* and *Bacilliarophyceae* occurred. In this study, no recovery was observed following chronic exposure to test concentrations above 0.1 mg/L and EC_{50} values for biomass reduction were less than 0.07 mg/L. Also, impacts on dissolved oxygen and phytoplankton were mirrored by similar concentration-dependant secondary effects (reductions in zooplankton abundance). A number of major taxa and total zooplankton populations were depleted with EC_{50} values < 0.6 mg/L. Recovery in many of the zooplankton taxa was observed at concentrations less than 1.0 mg/L.
Anderson (1981) has demonstrated dramatic reduction of periphyton diversity and density as well as depressed concentrations of dissolved oxygen following treatment of a small pond with hexazinone at 1 mg/L.

The impact of hexazinone in lake phytoplankton communities highlights the sensitivity of plants to hexazinone. As with most other forest-use herbicides, plant species are equally or more sensitive to hexazinone intoxication than other biota. Roshon et al. (1998) reported lowest median effect concentrations of 0.290 mg/L for inhibition of roots in an aquatic macrophyte *Myriophyllum sibiricum.* Schneider et al (1995) used similar experimental stream channels to examine short-term impacts on stream periphyton. They observed an 80% reduction in chlorophyll-a productivity during the VELPAR L addition and a recovery to control levels within 24 hrs with an estimated 4 h EC_{50} value estimated for chlorophyll a reduction as 0.0036 mg/L. Mean periphyton biomass and several metrics relating to potential macroinvertebrate effects were not significantly affected. The authors concluded that biota in flowing systems are resilient to short-term exposures to hexazinone. Kreutzweiser et al (1992) reported 1 hour LC_{50} values for more 13 aquatic insects from 5 different orders to be greater than 70 mg/L. Lethality and behavioral effects of hexazinone (Velpar L) on macroinvertebrates in outdoor stream channels (Kreutzweiser *et al.* 1992) has been examined in experimental stream channels under field conditions. Six species of stream insects were exposed to hexazinone at concentrations up to a maximum of 80 mg/L. None of the stream channel treatments affected the survival of any test species. Five of the six test species showed no behavioral (drift) response to the herbicide treatments, while significant drift of one mayfly species (*Isonychia* sp.) occurred at 80 mg/L, but not at 8 mg/L. In contrast to the aforementioned studies, even the lowest concentration of 0.05 mg/L hexazinone significantly reduced net primary production of periphyton on rocks in studies reported by Day (1992). Data presented in Table 7, demonstrate that hexazinone is relatively non-toxic to fish species and that formulated products are significantly less toxic than technical material (LC_{50}=275 mg/L). Berrill et al (1994) examined the effects of low concentrations of forest-use pesticides on embryos and tadpoles of several frog species (*Rana catebeiana* (bullfrog), *Rana pipiens* (leopard frog) and *Rana clamitans* (green frog). The authors suggested that the jelly layer surrounding frog eggs provides protection since even the high test concentrations of fenitrothion (insecticide) hexazinone and triclopyr (applied as the ester) showed no effects on embryo hatching success. Hexazinone had no effects on embryos or tadpoles even at the maximum concentrations tested (8 day exposure to concentrations of 100 mg/L).

6.8 Environmental Fate and Persistence of 2,4-D

As a member of the phenoxyacetic acid class of chemicals, 2,4-D is one of the oldest synthetic herbicides in use. Developed by the Dow Chemical Company (now Dow AgroSciences) in the mid-1940s, 2,4-D still enjoys widespread use as a weed control agent in cereal crops, turfrass, pastures and non-crop land in many countries worldwide. Surprisingly, the phytotoxic mode of action for 2,4-D is imperfectly understood. As noted by Crafts (1961) this group of compounds affects almost every biological activity of a plant including DNA, RNA and protein synthesis (Cheng et al 1972; 1973). A number of reviews on 2,4-D are available including those of Norris (1981); Norris et al. (1983), Ghassemi et al. (1981) and Loos (1975). The USDA-FS (1984) review pertains specifically to uses of 2,4-D in forestry, noting that it in 1976 it was the most widely used herbicide in U.S. forestry with application rates ranging from 0.28 to 22 kg/ha. Currently, 2.4-D is not being applied within the AOC however, similar use rates would be expected..

In plants, 2,4-D has physiological activity that mimics that of the plant hormone indole acetic acid (IAA), affecting stem elongation, mature cell growth, meristematic growth, as well as leaf and fruit abscission. Although similar to natural auxin compounds, the synthetic analogues are chlorinated aromatic ring structures are less susceptible to degradation *in planta*. None-the-less, 2,4-D is degrade in plants by dealkylation, hydroxylation of the ring structure and by conjugation, particularly to glucose or aspartic acid (Loos 1975). The acid form of 2,4-D is moderately soluble in water (90 mg/L, whereas the water solubility of the dimethylamine salt is substantially greater (3×10^6 mg/L) and the butoxyethanol ester is essentially insoluble in water. As for triclopyr, ester formulations of 2,4-D penetrate plant cuticles and are effective as foliar applications, whereas amine salt formulations are readily absorbed through plant roots. Both forms are translocated to meristematic or active growing regions of the roots or shoots.

In soils, 2,4-D is known to be rapidly degraded principally by microbial activity but also via chemical hydrolysis. While volatilization is variable depending upon soil conditions and form of 2,4-D involved, photodecompositon I considered to play a universally minor role in the loss of 2,4-D from soils.

Microbial degradation of 2,.4-D in soils has been extensively reviewed by Loos (1975) and Ghasemmi et al. (1981) who indicate that the principal metabolite is dichlorophenol and that the entire structure may be ultimately degraded to form carbon dioxide, water and chlorine. 2,4-D is not persistent in forest soils (DT_{50} typically < 30 days; Torstensson 1975; Norris and Greiner 1967; Thompson et al. 1984; Plumb et al 1977; Altom and Strizke, 1973; Radosevich and Winterlin 1979) and while classified as mobile (Helling 1971), it is relatively well adsorbed in surface litter (Norris, 1981) and so rapidly degraded that leaching or runoff with surface water is seldom observed in the field (Stewart and Gaul 1977; Thompson et al. 1984, Norris 1981). Norris (1981) has reviewed the fate and persistence of 2,4-D in forest litter and soils, showing that it is rapidly but reversibly adsorbed to organic materials. Since many microbial species have the ability to use this molecule as an energy substrate, significant deleterious effects on soil microbial communities or their ecological function are not expected. In the study by Fletcher and Freedman (1986), laboratory studies with two leaf litter and one forest floor substrate showed that for herbicides 2,4-D, 2,4,5-T or glyphosate, residue concentrations greater than 50x those of normal field levels are required to reduce litter decomposition rates. Hoffman and Albers (1984) reported on potential embryotoxicity and teratogenicity of 42 herbicides, insecticides and petroleum contaminants to mallard eggs, found 2,4-D, glyphosate, and atrazine to be only slightly or nontoxic (i.e. LC_{50} values equivalent to 196 to 550 kg/ha)

In aquatic systems 2,4-D amine formulations ionize and ester formulations hydrolyze to yield the free acid or anion, which in turn is rapidly degraded by both microbial, photolytic and hydrolytic processes (Zepp et al., 1975, Aly and Faust, 1964, Averitt and Gangstad, 1976). Under certain conditions, 2,4-D may also be lost through volatilization (Muir, 1991). Research on the fate and impact of 2,4-D which is pertinent to forestry has focused primarily on lotic systems wherein fate is controlled almost entirely by physical processes of dilution, downstream transport and sorption, which differ radically from those in lentic systems. Norris (1967) reported relatively high (0.84 mg/L) 2,4-D residues in samples taken 1 hr post-application from a stream draining an oversprayed marsh. In this study, residues of 0.076-0.176 mg/L persisted for up to 10 days thereafter and the author cautioned that such situations may lead to relatively large inputs into streams. Aldhous (1967) analyzed ditch water in a forest regeneration site after spraying with nonyl ester of 2,4-D (2.87 Kg a.e. ha^{-1}) and reported concentrations of 1.5-2.0 mg/L during the first 7 days, declining to non-detectable levels within 28 days. Maximum concentrations reported for 2,4-D in streamwaters of the Pacific Northwest range from 0.07 to 0.132 (Norris 1982, Norris and Moore, 1981).

As for other herbicides, salmonid fish tend to be most senstive to 2,4-D and in a pattern similar to that for triclopyr, ester formulations of 2,4-D tend to induce greater mortality with the lowest LC_{50} values for fish are in the range of 1-10 mg/L. Field studies investigating the effects of 2,4-D on stream organisms have been reported. Lorz *et al.* (1979) exposed coho salmon smolts to 200 mg/L 2,4-D (DMA) for 6 d, and then marked and released them in a nearby stream. They reported no mortality of the salmon, nor did the sublethal concentrations inhibit or significantly affect the downstream migration of smolts. A mixture of 2,4-D and 2,4,5-T (butoxyethyl ester) was aerially applied to forests in Japan at 4.05 Kg/ha. An impact assessment was conducted in a stream flowing through the treated area, and demonstrated no mortality or unusual behaviour of fish (salmon, dace, species unknown), and no measureable effects on macroinvertebrate abundance (Matida *et al.* 1975). In non-flowing systems, the DT_{50} (7.8 days) and DT_{95} (~24 days) estimates reported by Solomon *et al* (1990) for aqueous concentrations of 2,4-D acid are within the range of values reported in many studies southern studies. Given the timing of the application (June 26), the low pH (4.5) and of water clarity typical of such acid lakes in northern Ontario, rapid dissipation through photolysis and sorption of the IOE ester might be expected. Initial aqueous residues (2,4-D acid) quantified on day 1 represented approximately 40-50% of expected nominal concentrations. A relatively large proportion of chemical sorbed to side-walls of the enclosures (probably ester form sorbed to periphytic biofilms) was noted and sorption to sediments appeared to increase with time. A summary of the general literature on 2,4-D and toxicity of other phenoxy herbicides may be found in NRCC (1978) which emphasized the relatively high toxicity 2,4-D esters. One LC_{50} value for 2,4-D IOE in fish (On*chorhynchus grobuscha)* approximated 1 mg/L and was similar to the value for the highly sensitive cladoceran *Daphnia magna*. In general, laboratory data on the toxicity of 2,4-D to aquatic organisms, particularly zooplankton and fish, demonstrates the important influence of formulation on toxicity. Formulations used in forestry (DMA and IOE), are relatively less toxic than other forms of the compound. Zooplankton, phytoplankton and vascular plants appear to be most sensitive to the toxic effects of 2,4-D. Roshon et al. (1998) reported lowest median effect concentration of 0.007 mg/L for 2,4-D acid for inhibition of root growth in *M. sibiricum*. As with triclopyr, it is reasonable to expect that slightly acidic conditions, cool temperatures and low light intensity which may enhance persistence of the ester may also enhance uptake and exposure by aquatic organisms, resulting in greater impact.

6.9 Environmental Fate and Persistence of Imazapyr

Imazapyr is a member of the imidazolinones, which is one of the newest classes of herbicidal compounds, introduced and registered by American Cyanamid. Imazapyr is a broad-spectrum postemergence herbicide with residual soil activity and use patterns including weed control forestry, rights of way, and on industrial sites. Owing to its relatively recent introduction, there is a limited use history for imazapyr in North American forestry. A 1987 draft risk assessment document prepared for the southern USA indicates imazapyr applications on a total of 1600 acres, with projected increases of up to 6,500 acres. Consistent with this class of herbicides, imazapyr is used at lower rates (0.72 kg/ha) than many conventional herbicides (Cyanamid, undated).

In plants, imazapyr is readily absorbed through foliage and roots (Nissen et al 1995; Little et al. 1994, Tucker et al. 1994) and translocated rapidly throughout the plant in both xylem and phloem (Chamberlain et al. 1995). The phytotoxic mode of action for imazapyr depends upon binding to the plant-specific enzyme acetohydroxyacid synthase (AHAS). Enzyme inhibition disrupts synthesis of branched-chain amino acids (valine, leucine and isoleucine) (Shaner et al., 1984) ultimately resulting in impaired protein and DNA synthesis, cell growth and death. Imazapyr has a low volatility and thus is unlikely to be lost from treated plant surfaces by volatilization but it is highly susceptible to photodegradation. It is rapidly absorbed through plant foliage (87-94% in 2 hrs) and thus not susceptible to rainwash from treated surfaces where rainfall occurs beyond 2 h post-treatment (Cyanamid, undated).

In soils, imazapyr is degraded slowly by aerobic microbial as well as physical processes (Cyanamid, undated) and is variously persistent with DT_{50} values ranging from 90 days to 3years (Worthing and Hance, 1991). . Michael (1986 as cited in Cyanamid 1988) reported on the fate of imazapyr following aerial applications to watersheds in Alabama. DT_{50} values ranged from 19 to 34 days for imazapyr in forest soils and 37 to 44 days in litter typical of the southern USA. Most offsite movement occurred in the first two storm events following application but was a result of surface flow rather than leaching based on only 1% of total imazapyr observed in soils below 30 cm depth. Curran et al (1992) investigated photolysis of imidazolinone herbicides in aqueous solution and on soil and concluded that photodegradation could be an important loss mechanism in the field particularly on coarse-textured, moist, soils. Conflicting results concerning imazapyr leaching and lateral movement have been published. In field studies with radiolabelled imazapyr applied to a sandy loam soil in New Jersey (Cyanamid, undated) 80 to 90% of the residue remained in the top 15 cm of soil at 1 year post-application, suggesting that limited vertical or later mobility had occurred, other studies on accidentally flooded sites were cited supporting this contention. Although it is moderately well adsorbed (Kd 1.7-4.9) to soils and described by Worthing and Hance (1991) as not susceptible to leaching, some studies suggest that imazapyr is mobile (Mangels, 1991) and quite susceptible to leaching (Wehjte et al., 1987; Vizantinopoulous and Lolos, 1994; Weeks et al., 1987). Thin-layer soil chromatography studies indicate that sulfometuron was slightly more mobile than imazapyr (Wehjte et al. 1987). Imazapyr was very poorly sorbed to all but one of the 5 soils tested in laboratory experiments, with maximal sorption of 29% on a sandy clay loam soil. These studies also suggest that sorption of imazapyr is dependent upon minerology and that least adsorption occurs on soils of pH >5.8 and where moisture content >50% of field capacity. Vizantinopoulous and Lolos (1994) examined the persistence and leaching of imazapyr in clay loam and loam soils (OM3-4%, pH 7.75) using a bioassay technique and showed that 65 to 80% of imazapyr was leached from 50 cm soil cores with a total of 81 mm rainfall over a period of three days (14 mm/hour). DT_{50} values calculated from this experiment ranged from a maximum for the upper soil layer of 49.5 months to a minimum of 7.8 months for the 20-30 cm soil layer. Significant quantities of imazapyr leached down to the 45 cm depth confirming the mobility of imazapyr as previously described by Mangels (1991).Imazapyr is essentially non-toxic to earthworms (14 day LC_{50} 132.5 mg/kg) and birds (LC_{50} >2150; > 5000 mg/Kg for bobwhite quail and mallard ducks respectively) (Cyanamid, undated) suggesting a low very potential for effects on birds or soil organisms.

In aquatic ecosystems, imazapyr is degraded rapidly and extensively by photolysis with DT_{50} values of 1.9 to 2.7 days depending upon pH. A variety of degradation products were observed including complete degradation to form CO2. With LC_{50} values of > 100 mg/L for all species tested, neither the technical nor formulated product are substantially toxic to fish (Worthing and Hance, 1991). In the study by Michael (1986 as cited in Cyanamid1988), there was no indication that of off-site movement with stream sediments (aqueous residue concentrations were not provided). Field studies examining the biological effects of imazapyr on aquatic organisms are generally lacking.

Table 7. **Selected[1] toxicity data for major forest-use herbicides relative to freshwater fish and amphibian species.**

Herbicide	Organism	LC_{50} (mg/L)	Reference
Atrazine			
technical	*Salmo gairdneri*	4.5	Bathe et al. 1975
	Salmo gairdneri	3.5-5.7	Bathe et al. 1976
	Salvelinus fontinalis	6.3	Macek et al. 1976
	Lepomis machrocirus	>8.0	Macek et al. 1976
	Onchorhynchus kisutch	15	Lorz et al. 1979
	Rana catesbeiana	11.55	Birge et al. 1980
	Rana pipiens	22.89	
2,4D			
Isooctyl ester	*Salmo gairdneri*	70.3	Wan *et al.* 1986
(with 20% oil)		10.1	
Isooctyl ester	*Onchorhynchus gorbuscha*	1-5	Meehan *et al.* 1974
Isooctyl ester	*Salmo gairdneri*	~50	
Isooctyl ester	*Salmo clarki*	>60	Woodward and Mayer 1978
	Salvelinus namaycush	>60	
DMA	*Salmo gairdneri*	58.6	Wan *et al.* 1986
(with 5% oil)		35.4	
DMA	*Lepomis machrocirus*	100	Johnson and Finley 1980
	Pimephales promelas	335	
acid	*Salmo gairdneri*	>50	Meehan *et al.* 1974
	Oncorhynchus kisutch	~50	
	Salvelinus namaycush	45	Johnson and Finley 1980
	Salmo clarki	64	
Glyphosate			
ROUNDUP	*Salmo gairdneri*	1.4	Folmar *et al.* 1979
	Pimephales promelas	2.3	Folmar *et al.* 1979
	Lepomis macrochirus	1.8	Folmar *et al.* 1979
	Onchorhynchus gorbuscha	33	Wan *et al.* 1989
	Xenopus laevis	9	Perkins 1996
Acid	*Lepomis macrochirus*	24.0	Monsanto 1982
	Salmo gairdneri	106	Monsanto 1982
	Salmo gairdneri	10	Wan *et al.* 1989
	Onchorhynchus gorbuscha	14	Wan *et al.* 1989
	Xenopus laevis	5407	Perkins 1996
Triclopyr			
Acid	*Salmo gairdneri*	117	Dow 1983
	Lepomis macrochirus	148	Dow 1983
	chinook	2.9	Wan *et al.* 1987
TEA	*Pimephales promelas*	120	Mayes et al in USDA-FS 1984
	Salmo gairdneri	>100	Johnson and Finley 1980
	Xenopus laevis	163	Perkins 1996
TBEE	*Salmo gairdneri*	2.2	Servizi *et al.* 1987
	Pimephales promelas	2.3	Dow 1981

	Onchorhynchus gorbuscha	1.2	Wan *et al* 1987
	Salmo gairdneri	0.74-0.87	Dow 1981
	Onchorhynchus gorbuscha	0.5	Wan *et al.* 1987
	Rana spp. (Tadppoles)	1.2-2.4	Berrill et al. 1994
	Rana spp. (Embryos)	8.0 (NOEC)	Berrill et al. 1994
	Xenopus laevis	9	Perkins 1996
Sulfometuron-methyl	*Salmo gairdneri*	>12.5	
	Lepomis macrochirus	>12.5	
	Xenopus laevis	> 10.0 mg/L	Fort, 1998
Hexazinone			
VELPAR	*Lepomis macrochirus*	925	Schneider & Kaplan 1983
	Salmo gairdneri	872	Wan *et al.* 1988
	Onchorhynchus gorbuscha	676	Wan *et al.* 1988
	Rana spp. (Tadppoles)	>100	Berrill et al. 1994
	Rana spp. (Embryos)	>100	Berrill et al. 1994
PRONONE	*Salmo gairdneri*	1964	Wan *et al.* 1988
	Onchorhynchus gorbuscha	1408	Wan *et al.* 1988
technical	*Salmo gairdneri*	320-420	DuPont cited in USDA-FS1984
	Pimephales promelas	274	DuPont cited in USDA-FS 1984

1. Data selected to maximize pertinence to species of concern in AOC and to include most senstive as well as most commonly tested species to enhance comparability. LC_{50} data for fish species are for 96 hr exposures, while those for amphibia typically reflect exposures from spawning through 96 hours post hatching
2. Actual formulation tested was ROUNDUP however this formulation is identical to VISION.
3. Actual formulation tested was GARLON 4, however this formulation is identical to RELEASE.

7.0 Conclusions, Risk Analysis and Recommendations

7.1 Conclusions

Herbicides are applied to private forest land holdings of the Pacific Lumber Company (PALCO) for control of competing vegetation, roadside weeds and invasive exotic species such as pampas grass (*Cortaderia selloana*). The herbicides currently or potentially used within the designated area of concern include atrazine (AATREX), glyphosate (ROUNDUP and ACCORD), triclopyr (GARLON 4 and 3A), hexazinone (PRONONE), sulfometuron-methyl (OUST), 2,4-D (ESTERON99; LV4 and A1) and imazapyr (ARSENAL). All products are registered for such uses under both federal (Environmental Protection Agency) and state (California Department of Pesticide Regulations) legislation and are applied according to these laws and the specifications of the product label. This fact, in and of itself, strongly supports the concept that significant deleterious effects in the general environment or to common animal species are not to be expected. However, both abiotic and biotic variables peculiar to the AOC may generate unique environmental concerns, thus warranting a site-specific review. In this regard key issues which have been identified relate to:

Key Issues

a) the potential for herbicides to move off-site and contaminate aquatic environments
b) risks to amphibian species of special concern (red-legged, yellow-legged and tailed frogs as well as southern torrent salamanders)
c) risk to reptilian species of special concern (northwestern pond turtle)
d) risk to piscine species of special concern (coho salmon)
e) risk to avian species of special concern (yellow warbler, yellow-breasted chat)

Review of the scientific literature shows that an extensive database on environmental chemistry, fate and toxicology of these herbicides as derived from laboratory studies is available and sufficient to support general environmental risk assessment. Further the site to which the assessment pertains is reasonably well characterized and the natural histories of key species of concern are known. Finally, laboratory data is significantly augmented by the results of field studies conducted in forested ecosystems in the west coast of the USA and throughout North America which document both fate and effects of these herbicides under more realistic environmental scenarios. Information on fate and impact of the classical and most frequently used herbicides (atrazine, glyphosate, 2,4-D, triclopyr, hexazinone) are most comprehensive, those for more recently registered compounds (e.g. sulfometuron-methyl, imazapyr) are still developing. Toxicological information on avian and fish species is relatively extensive, whereas less data is available for amphibians. Pertinent data for reptiles and salamanders is largely unavailable as is information on potential synergistic or additive effects of herbicide mixtures. With respect to mixtures only one study (Abdelghani et al.1997) was found.

Based on a comprehensive review of the pertinent scientific literature a number of general conclusions can be drawn. All herbicides currently used or expected to be used within the AOC are moderately to highly water soluble (exceptions are esters of 2,4-D and triclopyr) and non-bioaccumulatory. They are subject to a variety of abiotic and biotic degradation mechanisms as well as several dissipation pathways, and are therefore generally non-persistent in plant, soil, or aquatic compartments. Compounds with greatest persistence in soils and vegetation are atrazine and imazapyr. Principal degradative metabolites of all herbicides have been identified and are generally more water soluble, more susceptible to metabolism and less persistent than their respective parent compounds. Herbicides are applied within the AOC by ground-based techniques only. Given the methods and frequency of herbicide application to any single site, the relatively low percentage of the land base to which they are applied (i.e. 2,000 acres per year or 5.8% per annum of total clearcut land proposed in the sustainable yield program), and the fact that herbicides are applied to burned sites which are marginal habitat for most species, direct exposures to wildlife are expected to be improbable and infrequent. The tendency for mobile species to avoid humans or seek cover upon intrusion into their habitat further reduces the potential for direct exposure. Therefore, the principal exposure route for species in or near the treated sites is considered to be indirect through ingestion or dermal contact with environmental residues. Maximal residue levels are expected within the treatment area per se and particularly in target vegetation with some material depositing on the soil either directly or indirectly. Potential contamination of riparian or aquatic environments may occur through leaching or later transport with surface water movement. The risk for offsite movement via these mechanisms is greatest for runoff via ephemeral stream channels which may be oversprayed and where significant and intense rainfall events occur shortly following herbicide applications. In general, maximal residues expected in either vegetation, soil, or aquatic compartments of the environment are well below those which might be expected to generate direct, acute toxicological effects on wildlife. The general conclusions noted above are consistent with the findings of several previous reviews (Neary et al. 1993; USDA-FS 1984; USDA-FS 1987; Norris 1982; Serfis 1986). Several monitoring studies conducted within the AOC clearly and repeatedly demonstrate that herbicide residues above analytical limits of detection do not occur in Class I and II streams draining treated areas, strongly supporting the contention that there is minimal risk to organisms inhabiting these environments. Risk assessment relating to specific key issues for the AOC are summarized below.

7.2 Risk Assessment

With respect to the site-specific key issues the following conclusions regarding potential risks can be drawn:

a) the potential for herbicides to move off-site and contaminate aquatic environments

Comparative physico-chemical properties, as well as results of both laboratory and field studies (**Risk Matrix 1**) demonstrate that neither glyphosate nor triclopyr pose a significant risk for contamination of streams via leaching or runoff, any amount of these chemicals entering streams via these mechanisms would be in the least toxic acid forms. If it were to be used within the AOC, 2,4-D might pose a moderate risk of stream contamination. However, owing to its rapid degradation in soils, 2,4-D is not considered susceptible to leaching and would only be mobililized to any significant extent with surface runoff where rainfall events occur shortly after application. Application of ester formulations of 2,4-D and triclopyr which have significantly higher binding affinity for organic carbon, is an important factor further mitigating against stream contamination potentials for these compounds. The risk of stream contamination by imazapyr is assessed as moderate, largely on the basis of limited or equivocal field study results. Although its physicochemical properties suggest a potential for leaching, field results from two different forestry studies fail to corroborate this. Similarly, field studies have shown conflicting results with regard to potential for surface runoff. Potential risk associated with this herbicide is obviated by the fact that it is not currently used within the AOC. Decisions taken to use this product should be made under an adaptive management and monitoring approach as described below. The greatest risk of stream contamination is associated with the triazine herbicides, atrazine and hexazinone. These compounds have physicochemical properties conferring leaching and runoff potential, which has been confirmed by several field studies conducted in forest environments documenting either leaching below 30 cm soil depth or runoff with surface water. The fact that hexazinone is not currently used within the AOC negates any potential for stream contamination by this compound and any decision to use this product must be taken with due consideration for its demonstrated mobility.

The potential for herbicides to move off-site via leaching or surface movement is a complex function of several variables including rate and method of application, timing of herbicide application relative to periods of highest or most intense rainfall, general soil characteristics and degree of moisture content at time of application, slope, surface and subsurface channelling, degree of vegetative or sorptive material remaining on the site and existence of vegetative buffers surrounding drainage streams. Conditions within the AOC which enhance the risk of offsite movement for susceptible herbicides such as atrazine, include slope (> 50% in many areas of constituent watersheds), the common occurrence of ephemeral channels on treated sites, periodic high rainfall (particularly November through March ; > 9 inches per month), and the overlap of months with both high rain and high herbicide treatment frequency, particularly February and March. Factors which mitigate against risk of offsite movement include; targeted application, vegetation, large woody debris and organic carbon resulting from prescribed burning which remain on the site. Management practices which require retention of 170 and 100 foot vegetated riparian buffers about all Class I and II streams are particularly important mitigation mechanisms. Given the relatively high organic matter content, cation-exchange-capacity and fine-texture of soils within the AOC, substantial movement of atrazine via leaching is not expected. However, the use pattern for this herbicide, which typically involves applications during February, March and April when rainfall is frequent, and the fact that it may be applied to or near ephemeral stream channels may increase the risk of runoff losses. In 1998, the company began a replacement program whereby sulfometuron methyl was applied in place of atrazine. Based on the analysis presented in **Risk Matrix 1**, and considering its substantially lower use rates as compared to atrazine, sulfometuron methyl is not expected to contaminate streams. This postulate is supported by the studies of Wauchope et al. (1990), Michael et al (1987), Lym and Swenson (1991) and Wehtje et al. (1987) which all provide evidence that under environmental conditions pertinent to this assessment, sulfometuron methyl has little potential for movement in soil water. Several monitoring studies conducted collaboratively with the North Coast Regional Water Quality Control Board (NCRWQCB) at stream sites proximal to treatment areas have specifically addressed risks of stream contamination. These studies have repeatedly found non-detectable residues of glyphosate, triclopyr, sulfometuron-methyl, atrazine and diesel fuel under both baseflow and stormflow conditions in Class I and II streams. These results provide strong evidence that the combination of best management practices currently employed within the AOC, including the restriction for ground applications only, employment of 100 and 170 foot buffer zones about Class II and I streams respectively, incorporation of tracer dyes to enhance visualization during and after applications, supervision of herbicide applications by trained staff, effective site mapping and on-site control of contract applicators, mitigate the risk of contamination of Class I and II streamwaters to insignificance. No monitoring information documenting residues in ephemeral stream channels or in sediments were available.

b) ***risks to amphibian species of special concern (yellow-legged frogs, red-legged frogs, tailed frogs, southern torrent salamanders)***

Characterization of the exposure risk for the various amphibian species of special concern within the AOC is species and behaviour dependent. Since yellow-legged frogs are seldom found far from small, permanent streams, exposures would be predicated on contamination of Class I and II stream systems. The results of several monitoring studies provide strong evidence that this does not occur and supports the contention that yellow-legged frogs are not exposed to significant levels of any herbicide currently used within the AOC. This includes atrazine for which the risk potential is considered greatest . Further, results reported in the literature for field studies involving direct overspray scenarios show maximal residues which are substantially lower than median lethal toxicity values for frogs **(Risk Matrix 2)**. Finally, these data indicate that the toxicological risk potential is greatest for those herbicides (glyphosate and triclopyr) which, owing to their physico-chemical properties, have the least potential to contaminate streams.

For tailed frogs, which are highly specialized to live in swift, perennial streams with low temperatures (Nussbaum et al. 1983), the risk of toxicological effect is also considered minimal. Again, contamination of Class I and II streams has not been demonstrated by on-site monitoring programs. Further, even maximal residues of forest herbicides are substantially below median lethal toxicity values for various frog species. Thus, the risk of direct lethality to tailed frogs within the AOC, where herbicide concentrations are below detection limits, is exceedingly low. In this case, the fast-flowing stream habitat preferred by this species, provides an additional mitigating factor since several studies show that durations of exposure in such systems (1-2 h) are significantly shorter than typical laboratory exposure periods (48-96 hr).

Southern torrent salamanders habitat preferences are for cool, moist, areas of late seral stage forests. However, it is possible that even within cutover sites, southern torrent salamanders may inhabit seeps or springs containing flowing water. Since these potential habitats are designated as Class II watercourses and afforded appropriate protection measures (i.e. buffers of 100 feet) there is little chance for direct exposures of southern torrent salamanders. As these salamanders are rarely found more than one meter from water (Anderson 1968, Nussbaum and Tait 1977), indirect exposures are also highly unlikely. The improbability of exposure, either directly or indirectly, results in a low risk potential for southern torrent salamanders. However, no directly pertinent toxicological data for salamanders are available and the possibility that salamanders are unusually sensitive to herbicide intoxication does therefore exist.

Treatment sites, which are typically recently cutover and burned, represent extremely marginal habitat for red-legged frogs. However, anecdotal observations of red-legged frogs on herbicide treatment sites indicates that such exposures are not impossible. Habitat preference, low herbicide application frequency (2 out of every 60 years), low total percentage of the landbase treated annually and the fact that treatment areas are non-contiguous are all elements inherently mitigating against the risk of direct exposures to red-legged frogs within the AOC. Also, since herbicides used within the AOC are applied by ground application techniques typically involving teams of backpack sprayers and since biota have a natural tendency to avoid humans, these amphibians would be expected to move away from applicator teams and seek cover as they pass through the area, further reducing the potential for direct exposure. Data presented in Table 7, demonstrate that amphibians as a group do not appear to be unusually sensitive to herbicide intoxication. However, the potential for direct and indirect cutaneous exposures within treated areas confers a higher risk for this species as compared to other amphibians under consideration. For the herbicides which have been examined for toxicity to frogs (all except 2,4-D) hexazinone appears to be least toxic. Formulated products of glyphosate and triclopyr as well as atrazine all have LC_{50} values approximating 10 mg/L, however given differences in life stage, mechanism of exposures, and the vagaries of potential exposures in the field extrapolative risk is highly uncertain in this case.

c) risk to reptilian species of special concern (northwestern pond turtle)

The specific habitat of the northwestern pond turtle includes includes drainage ditches (Zeiner et al. 1990a, Bury 1962, Holland 1994, and Nussbaum et al. 1983). In addition, this species is known to move hundreds of meters away from aquatic areas and nest in terrestrial sites. Thus, there is a potential for these turtles to be directly exposed to herbicides either on terrestrial treatment sites or when herbicides are applied around ditches for roadside weed control. No toxicological information could be found upon which risk to reptiles could be evaluated.

d) risk to piscine species of special concern (Coho salmon)

Coho salmon migrate into freshwater tributaries in late fall and winter and lay eggs in swift flowing gravel-bottomed streams. Alevins and young typically inhabit such areas for up to a year following spawning. A comparison of literature values for maximal concentrations of herbicides observed in forest field studies with median lethal concentrations for the most sensitive freshwater fish species (**Risk Matrix 2**), suggests that only triclopyr and 2,4-D esters present a significant risk to coho salmon. Since these herbicides are applied by ground application only to treatment areas outside the 170 and 100 ft buffer zones protecting Class I and II streams respectively, potential direct contamination of streamwater is considered negligible. Field and laboratory studies (Bentson and Norris,1991; Thompson et al. 1994) demonstrate that ester forms of these compounds are rapidly hydrolzyed or photochemically degraded to the acid form upon release to the environment, indicating that triclopyr residues entering streams by runoff or leaching mechanisms would be in the least toxic acid form. Several studies have documented that realistic worst-case exposures regimes for triclopyr ester in flowing waters are of insufficient magnitude and duration to generate siginificant toxic effects in salmonid fish species (Thompson et al. 1991; Kreutzweiser et al.1995). In the case of coho salmon the low potential for sublethal or indirect effects is supported by several studies on the most toxic compound (triclopyr ester) which show no effects on aquatic macroinvertebrates comprising the majority of their diet (e.g. Mayes *et al.* 1989. Fontaine 1990; Kreutzweiser et al. 1995) and by the studies of Janz et al. (1991) and Kreutzweiser et al.(1995) which demonstrate no significant physiological stress or growth effects in sensitive salmonid species even following direct exposures. Given that several monitoring studies show non-detectable residues of triclopyr in streams within the AOC, there appears to be essentially no risk of either direct or indirect toxicological effects of either triclopyr or other herbicides on coho salmon.

e) *risk to avian species of special concern (yellow warbler, yellow-breasted chat)*
Since herbicides used within the AOC are applied by ground application techniques only, to only a very small percentage of the total landbase annually and that they are applied to non-contiguous regeneration sites on a temporal frequency of typically 2 out every 60 years, direct exposures to avian species within these systems must be characterized as improbable and infrequent. This is particularly true for the yellow warbler and yellow-breasted chat which nest (1-60 feet above ground) in riparian habitats. These birds would be expected to make minimal use of herbicide treatment sites and would be expected to move away from applicator teams and seek cover as they pass through the area, making direct exposures unlikely. Based on worst-case exposure **(Risk Matrix 2)**, indirect exposures resulting from herbicide contamination of insects and berries which form the bulk of their food is not considered sufficient to induce toxicological effects in most cases. As exceptions, atrazine and 2,4-D appear to have relatively higher potentials for effects on birds. However, for these species indirect effects through habitat alteration or food reduction are also unlikely since herbicides are not used directly in their primary foraging habitat. Several studies (MacKinnon and Freedman 1993; Slagsvold 1977; Morrison and Meslow 1984a&b; Santillo et al 1989; Hardy and Desgranges 1990) fail to demonstrate substantive population reductions in neotropical migrant birds during the growing season after treatment and show that changes in utilization of herbicide-treated sites is specific. Where species densities are reduced, they recover during the following growing season. Since these studies involve direct over spray of regenerating clear-cuts in which those species were nesting and foraging, it is unlikely that indirect exposures as would occur within the AOC would have substantial effects on yellow warblers or yellow-breasted chats.

Considering the ground-based methods and low frequency of herbicide application to any given site within the AOC, the non-persistent nature of the compounds, their general lack of tendency to move off-site (exception atrazine and hexazinone) and the existence of untreated 170 and 100 foot riparian buffers around all Class I and II streams including seeps and springs, the overall risk of direct toxicological effects to amphibians (yellow-legged frogs, tailed frogs, southern torrent salamanders) and fish (coho salmon) appears to be insignificant. This postulate is strongly supported by the fact that on-site monitoring studies have failed to show detectable residues for any herbicide in streams which are the preferred habitat for these species. Further, scientific literature indicates a greater than 2 fold margin of safety between maximally observed streamwater concentrations and laboratory median lethal toxicity estimates for herbicide-sensitive species combinations. Highest toxicological risks appear to be associated with the esters of 2,4-D and triclopyr and for ROUNDUP formulations which have the least potential to mobilize and contaminate these stream systems via leaching or surface runoff. The existence of 100 to 170 buffers vegetated buffers about these systems is a significant mitigating factor. The fact that no herbicides are employed within these buffer zones is protective not only of the aquatic habitat but also of species such as the yellow warbler and yellow-breasted chat for which riparian zones are preferred habitat. Under the conditions of herbicide use within the AOC, there appears to be no scientific basis upon which a significant risk to these bird species may be postulated. Of the key issues outlined for this assessment the greatest risk potentials may be for red-legged frogs and northwestern pond turtles. Although the potential for significant effects on these species also appear to be minimal, insufficient toxicological and exposure information results in an equivocal assessment of risk.

7.3 Recommendations

Based on the foregoing assessment, it appears that the risk of offsite movement of most herbicides and potential toxicological effects on most species of concern have already been mitigated to insignficance. As such current best management practices should be considered satisfactory and should continue. The minimal, equivocal, risks associated with potential runoff losses of atrazine, hexazinone and sulfometuron methyl in ephemeral stream channels, as well as the potential for herbicide effects on red-legged frogs and northwestern pond turtles could be better understood or further mitigated by:

a) modifying the current monitoring program to quantify both aqueous and sediment residues in ephemeral stream channels and at the confluence of these ephemeral channels and Class II drainage streams proximal to treatment sites (see proposed monitoring program outlined below)

b) undertaking a biological census program to determine the actual frequency of occurrence of red-legged frogs, northwestern pond turtles and possibly southern torrent salamanders in habitats such as ephemeral stream channels, road side ditches and seeps/springs within typical herbicide treatment sites (census to be biased to temporal patterns of herbicide treatment)

c) further research (by the scientific community) to fill data gaps associated with the toxicity of herbicides and herbicide combinations to amphibians and reptiles under realistic exposure regimes and to quantify the bioavailability of sediment-bound residues to these or surrogate aquatic organisms

7.4 Proposed Monitoring and Adaptive Management Program

In concert with the recommendations made above, the author proposes that the company continue its voluntary involvement in aquatic monitoring programs with the North Coast Regional Water Quality Control Board (NCRWQCB) but that the focus of this program be changed to more directly examine runoff potentials in ephemeral stream channels and to quantify exposures in critical habitats for red-legged frogs and northwestern pond turtles. As both of these species are highly interactive with sediments, herbicides often partition preferentially to sediments and are more persistent in this matrix, it is important to quantify both aqueous and sediment phase residues, to effectively characterize exposure potentials. Key elements of such a monitoring program would include:

Ephemeral Channels

a) monitoring runoff water and sediments from oversprayed (spray allowed to directly enter the ephemeral channel proper), buffered (nominally 10 feet on either side of channel proper) and non-buffered (spray allowed up to channel edge, but not into channel proper) ephemeral stream channels, where ephemeral channel pairs are selected to maximize similarity

b) monitor water and sediments in small pools at the confluence of ephemeral channels and Class II streams draining treated sites, pools with minimal flow and depth to be selected

c) monitoring to be conducted both during applications and during the first two significant storm events following treatment, each sample to be comprised of a pool of three or more subsamples to maximize representation of average concentrations during each sampling event

d) monitoring to be conducted on three or more treatment sites where the more mobile herbicides -atrazine, hexazinone (if used), and sulfometuron are applied. Site characteristics including slope, sampling location, rates and times of application, soil moisture conditions, total rainfall and ephemeral channel flow rates should be fully characterized

Roadside Ditches

a) monitoring both aqueous and sediment residues in buffered and non-buffered sections of ditches proximal to operational herbicide applications made for roadside weed control

b) monitoring to be conducted at three or more locations fully characterized in terms of water depth, volume, herbicides and rates of application, width of buffer

c) several subsamples of water and sediments to be taken from transects along both buffered and non-buffered sections of the ditch and pooled to provide a representative sample of each matrix for analysis

Finally, it is recommended that results of the proposed monitoring program be used to adaptively manage herbicide applications such that operational flexibility is optimized while potential risks of offsite movement or toxicological impacts to species of concern are minimized. For example, in a scenario where buffers on ephemeral stream channels clearly reduce offsite movement and contamination of Class II streams, further monitoring and adaptive management would be required to establish the minimum effective buffer zone size. The company may also wish to consider the potential to shift the timing of herbicide applications slightly to maximize the number of acres treated between April and October when rainfall frequency is least

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Risk Matrix 1. Potential Off-target Movement of Herbicides Via Leaching or Runoff

Herbicide (kg a.i/ha.)	Rate[a] Soil	DT_{50}[b]	Koc[c]	GUS[d]	Leaching[e] Potential	Runoff[f] Potential	Stream[g] Monitoring	Risk of[h] Stream Contamination
Atrazine	4.4	112	90	4.2	+++	+++	---	High
Glyphosate	0.64	30	30000	-0.7	---	---	---	Low
Triclopyr	2.2	47	27	4.3	---	---	---	Low
Hexazinone	1.7	30	0.6	6.2	+++	++-	NA	High
Sulfometuron methyl	0.17	10	171	1.77	++-	---	---	Moderate
2,4-D	2.75	28	20	3.91	---	+++	NA	Moderate
Imazapyr	2.2	90	2.9	6.9	--	+-	NA	Moderate

a. Use rates expressed in kg/ha (active ingredient) are based on average use statistics in the AOC or typical use rates in forestry as observed in the literature (e.g. for hexazinone (Neary et al. 1983); for Imazapyr (Michael et al. 1986)
b. Average DT50 values as reported by Neary et al. 1993
c. Average Koc estimates as reported in various literature sources (principally Worthing and Hance, 1991)
d. GUS = Groundwater Ubiquity Score = log DT50 x (4-log Koc) (Gustafson 1989)
e. Examples of forestry field studies clearly demonstrating leaching below 30 cm soil depth (+ for each study showing leaching)
f. Example of forestry field studies clearly demonstrating surface runoff (+ for each study showing substantial runoff)
g. Results of monitoring studies which demonstrate no detectable residues in streams of the AOC (- for study with non-detectable residue)
h. Overall risk estimate for stream contamination by forest herbicides moving via soil water or runoff. High GUS (>4), + leaching potential, + runoff potential, + monitoring result would yield high risk estimate

Risk Matrix 2. Environmental Risk Assessment Matrix for Toxicological Effects on Birds, Fish and Amphibians

Herbicide	Rate[a] (kg a.i/ha.)	Birds[b] Exposure	Birds[b] Toxicity	Birds[b] Risk	Fish[c] Exposure	Fish[c] Toxicity	Fish[c] Risk	Amphibians[c] Exposure	Amphibians[c] Toxicity	Amphibians[c] Risk
Atrazine	4.4	1377	2000	High	0.0042	3.5	low	0.0042	0.41	low
Glyphosate										
Acid	0.64	200	4640	Low	0.27	10	low	0.5	5407	low
ROUNDUP	0.64	200	NA	NA	0.27	1.4	low	0.5	9	mod
Triclopyr										
Acid	2.2	689	1698	Low	0.35	2.9	Low	0.35	NA	NA
GARLON4	2.2	689	1923	Low	0.35	0.5	High	0.35	1.2 to 9	mod
GARLON 3A	2.2	689	3176	Low	NA	NA	NA	NA	163	NA
Sulfometuron methyl	0.17	53.2	5000	Low	0.044	12.5	Low	0.044	10	Low
Hexazinone	1.7	532	2258	Low						
PRONONE	1.7	----	------	-----	0.442	275	Low	0.442	100	Low
2,4-D										
Acid	2.75	861	472	High	0.84	45	Low	0.84	NA	NA
Esters	2.75	861	5000	Low	0.84	0.5	High	0.84	NA	NA
DMA	2.75	861	5000	Low		35				
Imazapyr	2.2	158	2150	Low	NA	100	NA	NA	NA	NA

a. Use rates expressed in kg/ha (active ingredient) are based on average use statistics in the AOC or typical use rates in forestry as observed in the literature (e.g. for hexazinone (Neary et al. 1983); for Imazapyr (Michael et al. 1986)

b. Bird exposures calculated based on ratios of 313 mg/kg per kg/ha a.i. applied derived from ground foliar applications reported by Thompson et al (1994) and assumed equivalent to concentration in vegetative food material. Exposure concentrations are compared to concentrations in feed which induce an LC50 response in birds under laboratory toxicity testing protocols (typically 5 to 8 day exposure durations).

c) fish and amphibian exposures (mg/L) were assumed to be equivalent to maximal aqueous residues observed in forest stream environmentsand constant for a period of 96 hrs to facilitate comparisons to minimum LC50 values available in the literature

d) Risk was estimated as low (ratio of toxicity:exposure = 10 to 100), moderate (ratio of toxicity: exposure >2 to 10) or high (ratio of toxicity:exposure <2)